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Eighty Years of Change:
The Montane Vegetation of Jasper National Park

by

Jeanine Marie Rhemtulla

A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of Master of Science

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ABSTRACT

Changes in vegetation patterns from 1915 to the present in the montane ecoregion of Jasper National Park, Canada, were examined. Repeat photography of a series of 1915 survey photographs was completed. Vegetation in the paired images was analyzed both qualitatively and through the development of a new quantitative method for interpreting oblique photographs. Maps of vegetation cover in 1949 and 1991 were constructed from aerial photographs, overlaid and analyzed with G.I.S. software. Results indicate a shift towards late successional vegetation types and an increase in crown closure in coniferous stands. Grasslands, shrub, young tree growth, and open forests have decreased in extent, and closed canopy forests have become more prevalent. Changes in human activity, including interventions in the fire regime, are likely largely responsible. The results of the work may help to define historical reference conditions and to help establish restoration goals for the montane ecoregion of the park.

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CHAPTER 1

Introduction

On the east side [of the Athabasca valley], the country at once opens into a wide and boundless prairie – the land of buffalo, and the hunter's paradise.

Alexander Ross, spring 1825

And at this time we didn't have all the jack pine there is now – it was more open. Now, this undergrowth and things – that comes – between them days and now. You see the spruce and jack pine are quite tall now. If they once catch fire, well you can't do very much with them...

Edward Moberly, 1980
(forcibly relocated from Jasper in 1909)
(Murphy 1980)

1.1 Background

There is growing recognition that the vegetation in the montane valleys of Jasper National Park, Alberta, may have changed significantly over the last eighty years. Evidence from historical photographs and written materials suggests that at the turn of the century, grasslands and open forests dominated the main valleys; today, many of these areas are covered with closed canopy coniferous forests. Shaped for millennia by the regenerative forces of fire, flood, and wind, it seems that the 20th century introduction of fire suppression and prevention, forced dislocation of aboriginal peoples, and onslaught of modern human activities may be affecting the composition and structure of vegetation communities in new ways.

Much of the debate in Jasper centers on the montane ecoregion. Occupying the lowest elevations of the park, it is located in the valley bottoms of the Athabasca River

and its associated tributaries. Warmer and drier than other regions of the park, the montane ecoregion supports a high degree of biological diversity (Holland and Coen 1982b). It provides critical reproductive and winter habitat for many species of wildlife, and serves as a travel corridor for large carnivores and ungulates (Holland and Coen 1982a). The characteristics that render it desirable for wildlife, however, also make the montane the prime destination for human visitors to the park. High quality habitat once home to grizzlies and wolves today hosts highways and hydroelectric power lines, golf courses, hotels and other tourist attractions. Although it constitutes a mere 6.9% of the park landbase, the montane ecoregion attracts the great majority of tourist visitors.

Direct habitat loss, the introduction and spread of non-native plants, and the displacement of wildlife from areas of heavy human use are among the consequences of increased human activity in the park. Perhaps more serious, however, are the more indirect changes in natural processes that have occurred. For example, the berming of the Snaring River during the construction of the railroads in 1911, may have eliminated the periodic flooding that helped to shape the vegetation along its banks. Analysis of fire history data suggests that fire occurrence and extent during the last century in the park were dramatically lower than in previous centuries (Tande 1979, Van Wagner 1995). Reasons for this are debatable, but include increased efficiency of fire prevention and suppression, climate change, and the forced dislocation of aboriginal peoples who may have used fire as a management tool (Heinselman 1975, Kay and White 1995).

Whatever the precise reasons for the changes in natural processes shaping the park landscape, many people are concerned about the resulting ecological effects. In Banff National Park, for example, there has been a decline in grasslands, shrublands, deciduous forest, and young coniferous forests over the last 50 years (Achuff *et al.* 1996). Older pine and spruce forests have become more common, resulting in decreased habitat diversity over the landscape as a whole.

While changes in the abundance and distribution of vegetation types are expected in a wild landscape, given the effects of natural disturbance and forest succession, many ecologists are worried that the changes that have occurred over the last century are straying outside the bounds of historical variation (Achuff *et al.* 1996). Park managers are therefore increasingly looking to such actions as the reintroduction of fire into the

landscape, both through carefully managed wildfires, and the setting of prescribed burns. The idea is to emulate historical processes, and therefore patterns, on the landscape. Unfortunately, although we have relatively good knowledge of current vegetation cover in the park, our knowledge of conditions in the past is extremely limited.

With this in mind, I began, two and a half years ago, a project to examine how the vegetation in the Athabasca valley has changed since the creation of the Park in 1914. Armed with a remarkable collection of historical survey photographs taken in 1915, and the first series of airphotos flown in the park in 1949, I pieced together snapshots of the vegetation in the valley at several dates. Changes in vegetation structure and composition were analysed with Geographical Information Systems (G.I.S.) software. The results provide the first detailed documentation of vegetation change in the montane ecoregion of Jasper National Park, and can be used to help set goals for future management of the area.

1.2 Vegetation of the Montane Ecoregion

Jasper National Park is located in the Rocky Mountains of Alberta, Canada (see Chapter 2, Section 2.2 and Figure 2.1 for detailed description and map). The Montane Ecoregion ranges in altitude from the lowest park elevations at 1000 m to about 1350 m; the upper boundary varies depending on aspect (Holland and Coen 1982b). Although lodgepole pine (*Pinus contorta* var. *latifolia* Engelm.[1]) is the most common canopy-forming tree species, the ecoregion is defined by the occurrence of Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), white spruce (*Picea glauca* (Moench) Voss), and trembling aspen (*Populus tremuloides* Michx.), interspersed with grasslands (Holland and Coen 1982b). A total of 34 vegetation types have been described in the ecoregion, many of which differ only in understorey composition (Holland and Coen 1982b). On a soil-moisture continuum, xeric sites are occupied by juniper scrubland and *Koeleria-Calamagrostis* grasslands, xeric-mesic sites by Douglas-fir woodland, white spruce forests are found in mesic areas, black spruce (*Picea mariana* (Mill.) BSP.) forests and fens in mesic-hygric sites, and hygric sites are occupied by *Salix-Carex* fens (La Roi and

[1] Latin names after Moss (1983).

Hnatiuk 1980). Lodgepole pine occurs throughout the wooded area of the montane ecoregion (La Roi and Hnatiuk 1980).

Scattered grasslands occupy small areas of the Athabasca valley bottom and lower south-facing slopes in the montane. Stringer (1973) classified the majority of these grasslands as the *Koeleria cristata-Calamagrostis montanensis* type. Characterized by sparse plant cover and low species diversity, it was reported as the most xeric, undeveloped, and nutrient-poor of all grasslands in Jasper, Banff, and Waterton Lakes National Parks (Stringer 1973). The formation of this grassland may in part be due to overgrazing, and with decreased grazing pressure, a more mesophytic grassland type might develop. A *Stipa richardsonii* - Shrub Savannah type has also been described for the park and prevails on moister sites (Stringer 1973).

Lodgepole pine is perhaps the most abundant and widely distributed tree in the montane. This quintessential "fire tree" establishes rapidly following forest fires, and is often the dominant tree in stands burned within the previous 150 years (Cormack 1953, La Roi and Hnatiuk 1980). Lodgepole pine is shade-intolerant and usually viewed as a seral species. On xeric sites where spruce and Douglas-fir recruitment is low, however, lodgepole pine regeneration may occur in the understorey thus forming self-replacing stands (Holland and Coen 1982b, La Roi and Hnatiuk 1980, Stadt 1993). Stadt (1993) reported two such sites in Jasper (one each in the montane and subalpine) while the Ecological (Biophysical) Land Classification for the park (Holland and Coen 1982b) included a lodgepole pine-juniper-bearberry vegetation type (C3) which is successionally mature. More commonly, however, lodgepole stands are succeeded by Douglas-fir on dry-mesic sites and by white spruce on mesic sites (La Roi and Hnatiuk 1980, Stringer and LaRoi 1970).

Douglas-fir occurs mostly on warm, dry-mesic sites on south-facing slopes, rocky ridgetops, and well-drained river terraces. Three intergrading Douglas-fir vegetation types have been described: forest stands found primarily in raised areas above the valley bottom, mixed coniferous savannah on exposed south-facing slopes and xeric flatlands, and open woodland on rocky ridges (Stringer and La Roi 1970). Douglas-fir vegetation types are successionally mature, ranging in age from about 100 to 300+ years old (Holland and Coen 1982b).

White spruce dominates on cooler, moister sites - north-facing slopes, ravine bottoms, and young river terraces (Stringer and La Roi 1970). As a shade-tolerant tree, it usually establishes itself under the lodgepole pine canopy, succeeding it as the pine senesces, although in some cases it may occupy a site immediately following a fire (vegetation types C26 and C37 in Holland and Coen 1982b). Pure stands are found mostly on alluvial flats along the Athabasca River, although scattered individuals are found throughout the area (Tande 1977).

Aspen stands occur sporadically across the montane landscape in moist to sub-xeric sites, usually on moderate to steep slopes (Holland and Coen 1982b). Pure stands are most commonly found on alluvial fans, river terraces, avalanche paths, and glacial till deposits (Lulman 1976). Aspen reaches maturity in 80-120 years, and although self-maintaining aspen communities can occur in some places, most stands require fire, flood or avalanche events to stimulate new suckering (Bartos and Mueggler 1981). In the absence of such regenerative processes, aspen is succeeded by white spruce on mesic sites, and by lodgepole pine and Douglas-fir on xeric sites (Lulman 1976).

Black spruce forests dominate lake margins, seeps, fens and peatlands (Laidlaw 1971), and hanging wetlands at higher elevations (Hettinger 1975) in the montane. On sites with infertile soils, pine and black spruce may co-occur following a burn; pine will soon dominate on xeric sites while black spruce will become dominant on wetter sites (Raup and Denny 1950).

As is now the growing consensus among ecologists, the successional pathways described above rarely continue to "completion" (Stadt 1993). As I shall explain in the following section, fire events and other natural processes are historically common on this landscape, continually interrupting succession and stimulating regeneration of pioneer vegetation

1.3 The Ecology of Change

Vegetation structure and composition is shaped by a number of different processes occurring at various spatial scales across the montane ecoregion. Factors such as avalanches, floods, wind, insect and disease outbreaks, and herbivory by large

mammals can substantially alter landscape patterns (Holland and Coen 1982b, Westhaver 1987, Achuff *et al.* 1996). Periodic flooding and avalanches, for example, are important for the regeneration of aspen stands on alluvial fans and steep slopes; these same stands may be negatively affected by high browsing pressure (Bartos and Mueggler 1981). Insect outbreaks are particularly common in older coniferous stands (Westhaver 1987). Windthrow and other processes may be especially significant at the stand level.

Fire is generally regarded as the major disturbance in the park, and much research effort has been devoted to understanding its effects. Most of the forests in the three-valley confluence area (centred on the Jasper townsite) originated following major fires in 1889, 1847, and 1758; most of the lodgepole pine forests in this area date to the 1889 fire (Tande 1979). Mean fire return intervals (defined as the average number of years between consecutive fires) varied from 17-26 years in the montane and 74 years in the subalpine between 1665 and 1975 (Tande 1979). The fire regime consists of both frequent low intensity fires and less frequent high intensity fires; burns equal to a fifth or third of the park area occurred one to two times a century between 1600 and 1889 (Van Wagner 1995).

Tande (1979) described four major types of fire-generated communities: even-aged stands, double-aged stands, multiple-aged stands, and stands which were scarred but lacked regeneration dating to the scar-date. Even-aged stands were most frequently found in the subalpine ecoregion and are characterized by less frequent fires of relatively high intensity. Longer fire-free periods in the subalpine ecoregion result in greater fuel accumulations, which may result in higher fire intensities once an ignition is successful. High intensity fires are more likely to result in extensive tree mortality, with subsequent establishment of even-aged stands (Tande 1977, Heinselman 1975). Double-aged stands (i.e. with two distinct age cohorts) were commonly found on the valley-bottom. A shorter fire cycle in these areas prevents high fuel accumulations, thus limiting fire intensity and reducing the incidence of stand mortality. Complex stands with multiple ages were found in the most rugged terrain, and result from the interplay of frequent low-to-medium intensity fires and physiographic gradients of moisture and fuel accumulation. Douglas-fir - grassland complexes were maintained by frequent low-intensity ground fires that reduced understorey growth and scarred trees without resulting in much tree mortality.

Studies of fire regimes in other Rocky Mountain parks have yielded varying results. In Banff National Park, montane vegetation appears to burn more frequently (21-56 year returns) than either lower (77-130 years) or upper (181 years) subalpine ecoregions (White 1984). Similar results have been reported for the Kananaskis Valley (Hawkes 1979). In Kootenay and Glacier National Parks and the Kananaskis Watershed, changes in the fire regime occurred in the mid-1700s; fire cycles before this date were typically shorter than they have been since (Masters 1990, Johnson *et al.* 1990, Johnson and Larsen 1991). The recent advent of longer fire cycles has been attributed to the cooler, moister climate of the Little Ice Age, which began in the mid-1700s and continued until the 1940s (Johnson and Larsen 1991).

The effects of human activity on changing fire regimes continues to be debated. Although it has been assumed that fire frequency increased during the period of European settlement (Byrne 1964, Nelson and Byrne 1966), recent studies suggest that the amount of burned area did not increase with the arrival of white settlers, railroad crews, locomotives, or tourists (Masters 1990, Johnson *et al.* 1990, Johnson and Fryer 1987, Johnson and Larsen 1991, White 1985, Van Wagner 1995). In Jasper, special patrol forces travelled daily along the railway line on velocipedes reporting any fires that might have been caused by the coal-burning locomotives (Canada 1914). In fact, fire has been negligible within the Jasper park area over the past seventy years (Tande 1979, Van Wagner 1995), a pattern which is also apparent in Banff, Kootenay, and Waterton Lakes National Parks, and Kananaskis Provincial Park (White 1985, Masters 1990, Van Wagner 1995, Barrett 1996, Hawkes 1979). No fire-free periods of this length have been found in the fire history records of any of these five parks (Barrett 1996, Van Wagner 1995, Hawkes 1979), leading researchers to question what has changed. The most common hypothesis is that fire suppression policies, introduced when the parks were established, have been extremely successful. The true efficacy of fire suppression, however, has been questioned. In many areas it appears that most large fires were stopped by rain, not firefighters (White 1985, Masters 1990).

Research into the sources of fire ignition is now underway. Both Banff and Jasper appear to lie in a lightning-fire "shadow", because many of the weather systems that create lightning ignitions pass over the area and strike further east in the foothills

(Heathcott 1996). Lightning fire occurrence in these parks does not appear to be sufficient to account for the historical fire regime. The missing ignition source may be First Nations peoples, who used fire as a management tool. Such use has been described in many Native groups who live in various ecosystems (Lewis and Ferguson 1988, Lewis 1980, Barrett 1981, Blackburn and Anderson 1993, Delcourt and Delcourt 1997). Metis settlers living in the Jasper park area in the late 1800's used fire to enhance forage and reduce disease in wildlife (Murphy 1980). Details of First Nations occupation in the park previous to that time is still uncertain, as is the overall extent and ecological importance of Native fires in the Rocky Mountains. Some researchers claim that aboriginal fire is responsible for much of the fire record (Kay and White 1995, Barrett 1981), while others maintain that aboriginal fire was only a small component of the total fire regime (Heinselman 1975, Johnson and Larsen 1991). The dislocation of First Nations peoples and the forced removal of Metis homesteaders from the parks could possibly have eliminated a major source of fire ignitions (but see Masters 1990). The great success of park fire policy may have been in preventing, rather than suppressing, fires.

1.4 A landscape of change

Whatever the cause for this long fire-free period, the ecological effects are becoming increasingly apparent. In the Banff-Bow Valley, where fire has been almost completely absent since 1936, the proportion of area covered by younger vegetation types such as herb, low-shrub, and young conifer forests has decreased greatly, and has been replaced by older, closed forest vegetation, particularly pine, spruce, and spruce-fir forests (Achuff *et al.* 1996). Most aspen stands are over 100 years old and declining in vigour. In the continued absence of fire, vegetation will not only become older, more forested and dominated by conifers, but one third of the vegetation types may be lost entirely from the landscape, resulting in a significant decrease in overall biodiversity (Achuff *et al.* 1996).

In Waterton Lakes National Park, aspen stands, Douglas-fir stands, and grasslands have been the most severely affected by the lack of fire (Barrett 1996). In ponderosa pine-Douglas-fir ecosystems in Southern Interior British Columbia, fire suppression

combined with grazing pressures and selective logging have also led to significant changes in vegetation composition: a 50% decline in open forest area and 50-135% increase in closed canopy forest at each of two landscape study areas (Taylor and Hawkes 1997). Vegetation changes between 1870 to 1982 in the Rocky Mountains in Montana included a decrease in early stages of early successional vegetation types, an increase in crown closure of dry forest types, and an increase in numbers of shrubs and trees on rangelands (Gruell 1983). Encroachment of forests into grassland areas in the absence of fire has also been documented in the Colorado Front Range (Mast *et al.* 1997), and the Rocky Mountains of Montana (Arno and Gruell 1986).

In Jasper, there has been a continuous decline in structural heterogeneity within coniferous stands over the past eighty fire-free years (Tande 1979), and an overall increase in crown closure of closed coniferous forests (Heinselman 1975). Contrary to evidence from other locations, little evidence of forest encroachment into grasslands has been found in Jasper during informal field reconnaissance (Stringer 1973, Heinselman 1975). Other ecological consequences of continued fire exclusion include: increasing forest stand-age; lack of stand regeneration; the eventual elimination of grasslands and savannahs; increased fuel accumulations; and changes in nutrient cycles and energy flows (Heinselman 1975). Long periods without fire, and consequent high fuel accumulations, may also change the fire regime of the montane from one dominated by frequent low-medium intensity fires to one characterized by less frequent, higher-intensity burns (Heinselman 1975, Barrett 1996).

1.5 Methods in Forest History Research

In the absence of long-term experiments or permanent plot data, historical information on past vegetation communities can be extremely useful in determining ecological responses to changing conditions (Veblen and Lorenz 1991). A number of different methodologies have been used to illustrate and analyze landscape change over time (Noss 1985) . Historical materials - journals, fieldbooks, and other early accounts of people travelling through the area of interest - can shed much light on past landscapes. A number of studies have relied heavily on such materials to reconstruct pre-European settlement vegetation in the Eastern United States (Nelson 1957, Bromley 1935). Caution

must be exercised when using historical accounts, however, because of potential bias, misrepresentation, and lack of systematic and comprehensive information (Noss 1985, Forman and Russell 1983, Schullery and Whittlesey 1995).

Historical photographs and the technique of repeat photography can be used to determine past ecosystem states. When an historical photograph depicting some feature of interest is found, the site from which the picture was taken is relocated precisely, and a repeat photograph is taken (ideally from the same angle with similar film type and lens focal length) (Rogers *et al.* 1984). The two shots are subsequently compared and the results are often quite dramatic. Care must be taken when reconstructing a landscape from such data since rarely do historical photographs depict a comprehensive coverage of the area of interest (Noss 1985). Although there have been many studies that have used repeat photography, (Rogers *et al.* 1984, Gruell 1983, Veblen and Lorenz 1991, Hastings and Turner 1965) analysis of change is usually limited to qualitative observations. Few studies have quantified landscape change observed in repeat photographs (Sinclair 1995, Webb 1996).

Historical materials prepared specifically to document the state of a given area lend themselves more easily to comprehensive and quantitative reconstructions of previous landscapes. For example, land surveys conducted to partition land among early settlers can be extremely valuable because in addition to describing general land characteristics, surveyors often documented the species, size, and age of a 'witness' tree, blazed every quarter mile. Many studies have used these data to compare past tree species distributions with current vegetation cover (Lorimer 1977, Johnson and Fryer 1987, Siccama 1971).

Old maps and aerial photographs have also been used to trace landscape change over time (Foster 1992, Callaway and Davis 1993, Thibault and Zipperer 1994). Methods usually include establishing a relevant classification system, interpreting the maps or photographs, and creating maps of landscape features that are representative of the time of data collection. In older studies, these maps were usually displayed side by side (as with repeat photography) for visual interpretation, and accompanied by simple analyses of changes over time, such as changes in percent cover of various classifications. More recently, researchers have used G.I.S. software to overlay these time-series maps and to

analyze the change between them (Bakker *et al.* 1994, Green *et al.* 1993, Knight *et al.* 1994).

Paleoecological methods such as the analysis of pollen taken from lake-bottom cores can also be used to reconstruct past vegetation states. Detailed unearthing and mapping of decomposing woody materials, has been used to piece together the history of single forest stands, while dendroecological techniques can be used to reconstruct changes in canopy density and disturbance history over time (Covington and Moore 1994, Veblen and Lorenz 1991). Finally, fire history studies and successional modeling can be used to backcast past vegetative states (Achuff *et al.* 1996).

1.6 Research Objectives

In order to understand fully the effects of nearly a century of vegetation change in the absence of fire in Jasper, historical data are needed. Aside from detailed data on fire history - process - in the park, however, few data exist on historical vegetation states - patterns - on the landscape.

The primary objective of this study was to analyse how vegetation composition and structure have changed in the montane ecoregion of Jasper National Park from 1915 to 1997. Specific hypotheses addressed were:

1. The abundance and distribution of individual vegetation types has changed from early successional vegetation types to later seral types such as closed canopy coniferous forest.

2. Forest vegetation has encroached onto previously non-forested areas such as grasslands.

3. Forest crown closure has increased.

4. The spatial pattern of vegetation is characterized by decreasing patch number and increasing patch size over time due to increasing homogeneity of vegetation at the landscape level.

The study was approached in two ways. First, I returned to the locations of several dozen historical survey photographs taken in 1915 and rephotographed the same views. I interpreted the vegetation in both sets of photographs and compared how the overall vegetation composition changed between 1915 and 1997. Transition matrices describing shifts in vegetation from one type to another during this time were also constructed. Details of this work are presented in Chapter 2. Secondly, standard airphoto interpretation techniques were used to develop vegetation maps for 1949 (the first aerial photographs flown in the park) and 1991. Changes in vegetation patterns across the landscape, and within-stand structure and species composition were analysed with the aid of G.I.S. software. A number of landscape metrics were also compared. Chapter 3 contains the results of this work. Finally, Chapter 4 contains an overall synthesis of vegetation dynamics in the montane ecoregion over the last eighty years including implications for future management of the park.

In addition to being one of the first studies to document historical vegetation change in Jasper, this work is one of the first to complete systematic, comprehensive repeat views of historical photographs in the Canadian Rockies, and one of the first anywhere to analyze such paired photographs quantitatively. Long-term goals for maintaining or restoring integrity of Jasper National Park can only be meaningful if they are grounded in historical knowledge of past ecological conditions (Higgs *et al.* 1998); this research is directed at increasing our understanding of the dynamic nature of this landscape.

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CHAPTER 2

Repeating the Bridgland Survey: Tracing vegetation change through historical photographs

2.1 Introduction

In June of 1915, M.P. Bridgland, a Dominion Land Surveyor, arrived in the Rocky Mountains of Alberta. He was charged with a formidable task – to supervise the creation of the first topographic map of the newly established Jasper National Park. He rose to the challenge: in the space of the next four months, he, his survey crew of five – an assistant, two horse packers, two cooks - and a team of pack horses set up 93 survey stations on mountain tops, cliff edges, and prominent points at ground level (Bridgland 1924). At each station, Bridgland and his assistant took photographs circling the entire horizon. Later that fall, these photographs, together with pages of theodolite measurements, were meticulously crafted into a topographic map covering 2300 km^2, almost a quarter of the park area (Bridgland 1924).

The Dominion Land Survey was likely pleased to have another piece of Canada mapped. Unknowingly, however, they had created a legacy that has long outlived the practical utility of that first map. The Bridgland photographs, 750 of them in total, have become an extremely valuable visual record of the state of the park in its early years. Systematically taken and comprehensive in coverage, they are unparalleled by any other early historical records in the area, and few in the Rocky Mountain region as a whole.

In the summer of 1997, armed with a large-format camera of my own, and a single assistant, I returned to a dozen of Bridgland's survey stations, and rephotographed the same views. Thus began a project to examine how the vegetation in the Athabasca valley has changed over the last eighty years.

The technique of repeat photography has been used in many places to illustrate landscape change over time. The basic methodology consists of rephotographing a subject, ideally from exactly the same point, with the same angle of view, and at the same time of day. The repeat photographs are then used to analyse the rate, nature, and

direction of change of the subject, to contemplate the causes for said change, and to create new visual records of the landscape for future use (Rogers *et al.* 1984).

The earliest use of the method has been credited to Professor Sebastian Finsterwalder, who used repeated photographic surveys to map glacier change in the eastern Alps in the late 1880s (Rogers *et al.* 1984). Since that time, ecologists, geographers, and geologists in particular, have used the method to investigate changes in vegetation and landforms. The classic work in the genre is the The Changing Mile (Hastings and Turner 1965), a study of the effects of human activity and climate on the plant cover in Arizona and adjacent Mexico. Similar efforts have been undertaken in the Rocky Mountains of Montana and Idaho (Gruell 1983), the Colorado front ranges (Veblen and Lorenz 1991), and Yellowstone National Park (Houston 1982). Each of these studies has relied on collections of historical photographs assembled from a number of different sources, taken by numerous photographers, for diverse reasons, and over a fairly wide time range. Occasionally, systematic and comprehensive views of a landscape are available, usually as a result of historical surveys undertaken for one purpose or another. For example, photographs of the 1891 Stanton Expedition, which surveyed the length of the Grand Canyon to document a possible railroad route, were used as the basis of a repeat study (Webb 1996). Although such an approach is only possible where this type of collection of photographs exists, it does afford a relatively unbiased photographic sampling of the landscape, thus eliminating one of the major critiques of the method of repeat photography (Rogers *et al.* 1984).

Perhaps the greatest difficulty with repeat photography is finding a way to adequately analyse the changes evident in the paired images. The majority of studies are based on qualitative, visual analysis of the repeat pairs. While many of the images are often dramatic enough to speak for themselves, quantitative analysis that summarizes the extent and direction of change is highly desirable. The difficulty, of course, is inherent in the geometry of oblique photographs: the scale of the picture varies throughout the image. Absolute calculations of area are therefore extremely complex. The few studies which have attempted quantitative analysis usually calculate relative or proportional, rather than absolute, measures of change on repeat images. For example, change in the size of sandbars in the Grand Canyon between 1889 and the present was calculated by

determining whether the sandbars had 'increased', 'decreased', or 'remained the same' in size from one picture to the next (Webb 1996). Relative change in tree numbers over time in the Serengeti was determined by counting trees in corresponding areas on matching photographs, and calculating instantaneous rates of increase – a relative measure which could be used to compare the results of individual pairs of photographs (Sinclair 1995).

In this chapter, I compare vegetation in the montane ecoregion of Jasper National Park between 1915 and 1997, through the repeat photography of the Bridgland survey images. A total of 53 images were rephotographed and analysed qualitatively. Twenty pairs were subsequently analyzed quantitatively, to determine relative extent and directions of change in landscape vegetation composition. It was hypothesized that: 1) forest cover and crown closure have increased throughout the study area; 2) forest has encroached on grasslands; 3) early successional vegetation types such as open forest and young regenerating tree stands have decreased in extent; and 4) anthropogenic activities have increased over the last 80 years. A discussion of the possible reasons for observed changes and implications for the management of the park follows.

2.2 Study Site

Jasper National Park (52 N, 118 W) is located in the Rocky Mountains of Alberta, about 400 km west of the city of Edmonton, and occupies a total of 10 880 km^2 (Holland and Coen 1982). The study site was located in the lower Athabasca River valley, in the central region of the park, north-east of Jasper townsite (Figure 2-1). It lay within the montane ecoregion, which occupies the lowest park elevations (±1000 to ±1350m asl). The study area extended from Jacques Creek and Vine Creek in the north to the confluence of the Maligne and Athabasca Rivers in the south. Boundaries of the study site along the east and west sides of the valley corresponded to the upper limits of the montane ecoregion. The study area was approximately 14 km long by 4 km wide, occupying a total of 64 km^2, or 8.5% of the total montane ecoregion within the park.

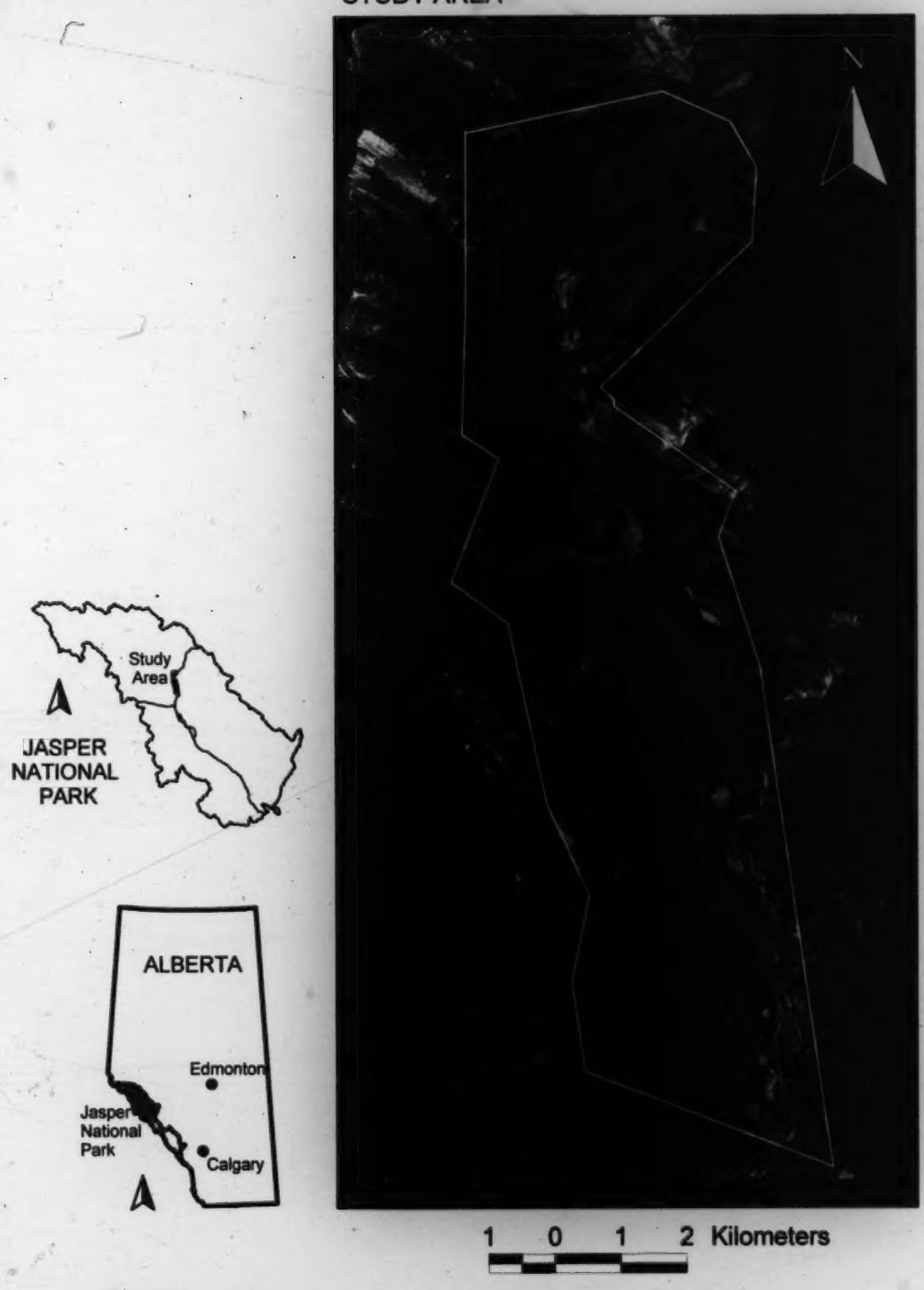

Figure 2 - 1: Study Area. White line represents the boundary of the study area in the montane ecoregion in the central area of Jasper National Park, Alberta,

The macroclimate of the park falls into the transitional zone between the Cordilleran and Continental climate regions of Canada (Seel and Strachan 1987). Regional climate is difficult to characterize because of both the tremendous physiographic influences that shape micro-climate, and the relative lack of comprehensive data. The climate of the montane ecoregion is generally the warmest and driest in the park, and has been classified as a Dfc climate (cold, snowy forest climate with no distinct dry season and cool short summers) (Holland and Coen 1982). The greatest temperature fluctuations are also experienced in the montane. Mean daily minimum and maximum temperatures are 7.3°C and 23.1°C in July, and –16.7 °C and -6.1°C in January, the hottest and coldest months respectively, as measured at Jasper townsite (Stringer and LaRoi 1970). Average annual precipitation in the montane is 471 mm (averaged throughout the montane ecoregion of the park) and 383 mm (measured in Jasper townsite) (Holland and Coen 1982).

The study area lies chiefly within the Front Ranges of the Canadian Rockies; the peaks are underlain by Late Palaeozoic limestone, and the valleys by Mesozoic shale (Gadd 1986). Surface material in the Athabasca River valley is primarily till and glaciofluvial deposits composed of sandstone and quartzites with some slate and limestone (Stringer and LaRoi 1970).

2.3 Methods

2.3.1 Data Acquisition

A general overview of the entire collection of Bridgland photographs was completed. Pictures which contained views that fell partially or completely within the study site were selected for rephotography. The general location of each survey station (to within about 500m) was obtained from the original topographical maps created by the Bridgland survey team. Original field notes with detailed survey information have been lost, and therefore precise camera locations were determined in the field by lining up copies of the original photographs with obvious landmarks on the landscape, using the principle of parallax (Rogers *et al.* 1984). Relocating precise camera stations was

extremely time-consuming; small camera movements, on the order of centimetres, could change an image significantly.

Camera equipment was chosen to approximate as closely as possible that used by the original survey team[1]. I used a Linhof Technika 4x5" large-format camera, 90 mm Linhof lens, and Manfrotto 055 tripod. All photographs were shot with T-max 100 (black and white) film. A No. 85C Wratten filter (pale orange) was used to cut haze and increase contrast. A total of 53 pictures were taken during the 1997 field season (see Appendix 1).

Film was processed commercially. Because the equipment used resulted in negatives with a slightly larger field of view than the originals, careful cropping of the images was necessary so that the repeats matched exactly both the size and field of view of the original set. Ilford RC Multigrade Paper and Ilford chemistry were used for printing and standard darkroom procedures were followed.

Qualitative analysis was based on visual inspection of all 53 pairs (1915/1997) of photographs. Twenty pairs of photographs were chosen for quantitative analysis (see Appendix 1). Views taken at ground level, which portrayed only very local vegetation, and at very high elevations, for which detailed identification of cover types was difficult, were eliminated. A vegetation classification system was developed based on the system used to interpret airphotos in a companion study (see Chapter 3). Attributes included physiognomic class (forest, open forest, shrub, herb, wetland, young tree growth, water, rock, sand/gravel, and anthropogenic sites), and crown closure for forest stands (A – 16-30% canopy cover, B – 31-50%, C – 51-70%, D – 71-100%). Forest stands were assigned to these cover classes based on a visual estimate. An attempt was made to differentiate between deciduous and coniferous forest types, however the results were not reliable for all repeat pairs so this aspect was limited to qualitative observations.

Photographs were covered with transparent acetate overlays. For each pair, 8-12 point features which could be accurately identified on each picture and which were well-distributed throughout the picture area, were selected as control points. Coordinates were obtained by laying the repeat (1997) acetate overlays on a cartesian grid. Areas of

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[1] Bridgland most likely used a large format camera, 4¾" x 6½" glass plate negatives, 164 mm Zeiss Tessar Series III lens, B/W panchromatic emulsion, and a Wratten and Wainwright "G" filter (yellow) (Bridgland 1924).

homogeneous cover (compositionally and structurally) were delineated with the aid of a 8x magnifying loupe. Minimum polygon size was approximately 0.5 cm^2. The historical and repeat photographs were interpreted as independently of one another as possible, in an attempt to minimise interpreter bias. Informal checking of polygon interpretation to ensure overall accuracy and to confirm uncertain polygons was done by comparing 1997 polygons to the corresponding area on 1991 airphotos (see Chapter 3). It was not possible to directly check the interpretation of the 1915 photographs.

Acetates were digitized, edited and labeled using GRASS Geographic Information Systems (G.I.S.) software (Shapiro *et al.* 1993). Accuracy of overlaying the two images in each pair (based on the selected control points) was assessed quantitatively using the residual mean average and qualitatively by observing how well obvious landscape features lined up. Coverages were subsequently imported into the IDRISI G.I.S. software for analysis (Eastman 1997).

2.3.2 Data Analysis

A spatial cross-tabulation was calculated for each pair of images[2]. These individual results were added to create a summary cross-tabulation for the entire study area. This approach assumes that a pixel unit is comparable both within and between photographs. In fact, because the scale of the images varies not only within a given image, but between the different survey photographs, adding picture pixels in this way biases the summarized data in favour of foreground pixels, and larger-scale photographs. A review of the photographs suggested no systematic bias that would favour one vegetation type over another, thus it was assumed that summarising data in this way would provide an adequate representation of the general relative trends in the study area.

A transition matrix illustrating directions of change between 1915 and 1997 was calculated from the summarised cross-tabulation data. For each cover type, the percentage of the total area of that cover type in 1915 moving to another type in 1997 was calculated (see Table 2-1 for detailed explanation). This provided an indication of the major trends of change in the study area. The extent of change in the overall landscape

[2] Cross-tabulation is a technique used to compare qualitative data in two matching images. Corresponding pixels in the images are compared and a table (transition matrix) listing the observed frequencies of every possible combination of the categories in the images is produced (Eastman 1997).

vegetation composition between 1915 and 1997 was estimated by summing the pixels in individual pictures by cover type at the two dates. The validity of this approach is subject to the same assumptions outlined above. It must be emphasized that these analyses do not provide absolute measures of change in the study area, but simply serve as estimates of the relative proportions of vegetation change that have taken place.

A more detailed explanation of methodologies employed is contained in Appendix 2.

2.4 Results

2.4.1 Qualitative analysis

A total of 53 repeat photographs were taken throughout the study area. Eleven of the pairs are reproduced here (Plates 2-1 to 2-11). They were selected to try and showcase as much of the study area as possible, and to represent the variation in the types of photographs that were taken by the Bridgland survey team. The observations presented here are based on an analysis of all 53 pairs.

The greatest apparent change in vegetation in the paired photographs was the dramatic increase in forest cover and crown closure throughout the study area (Plates 2-1 to 2-4). The 1997 photographs consistently showed older, more closed forest stands in the Athabasca Valley, and along the flanks of the mountains which surround it on both sides. There was no evidence of differential forest expansion on south vs. north facing slopes. There appeared to be a decline in the abundance of deciduous trees (Plates 2-2 and 2-3). Distinguishing between coniferous and deciduous species was difficult in many of the smaller-scale photographs, however, so this observation was based on a subset of the total photographs and may not be representative of the entire study area. There was also a decline in the number of young regenerating tree stands (Plates 2-1 and 2-4). Forest encroachment into former grassland areas was apparent, although several core grassland areas in the Henry House Flats region persist in the 1997 photographs (Plates 2-5, 2-6 and 2-11). Overall, the 1915 photographs depicted a valley with patchy vegetation – open coniferous forests stands, large grasslands, young tree regeneration, and the occasional stand dominated by deciduous species. The 1997 matching pictures suggested an increase

in the homogeneity of the vegetation cover – much of the former patchiness was replaced by uniform closed coniferous stands.

A number of the historical photographs showed evidence of burns (Plate 2-1). The extent of these was hard to discern as burn evidence was difficult to detect in many of the smaller-scale photographs. Entire burned stands were rare; more common were occasional standing dead (burned) stems interspersed with living trees in open forest stands. It is highly likely that many of the young regenerating coniferous and aspen stands apparent in the 1915 photographs were the result of fire events. There was little evidence of fire in the 1997 matching photographs. Burned stands resulting from park-sanctioned prescribed burns were apparent along the Colin Range (Plate 2-6) and in a small section of the Henry House grassland.

Changes were also evident in hydrological features in the study area. Retaining walls were built along the Snaring River, a major tributary to the Athabasca River, in the early 1910's to prevent flood waters from washing out the railroad bridges. Matching photographs suggested that these walls have contributed both to a narrowing of the river (and presumably a deepening of the channel) and to the replacement of the shrubby and deciduous vegetation along the river bank with coniferous tree stands (Plates 2-7 and 2-8). A small lake in the centre of the study area which had obvious above-ground connections to both the Athabasca and Snaring Rivers in 1915 appeared to be isolated from these river systems in 1997 (Plates 2-8 and 2-11). The wetland complex associated with this lake had also undergone some changes; low shrubs that covered much of the area in the historical photographs were replaced by herbaceous cover (Plate 2-8). The Esplanade wetland in the northern part of the study area seemed, superficially, to have undergone little change despite the numerous transportation corridors which now bisect it (Plates 2-9 and 2-10).

There was an overall increase in the human presence evident on the landscape. New human infrastructure included a borrow pit (Plate 2-2), a power generating station (Plate 2-3), campgrounds (Plate 2-8), and transportation and utility corridors (Plates 2-2, 2-6 and 2-9).

Plate 2-1: Athabasca River Valley, view N. (Station 57 – No. 459)

Original picture shows patchy vegetation in valley bottom. A recent burn is evident at bottom-centre (base of cliff), dense young coniferous growth at centre-left, and open coniferous forest mixed with aspen at lower-right. Dramatic increase in both coniferous tree cover and density is obvious throughout the retake – on cliff-ridge, valley bottom, and flanks of Colin Range (right). This and the next three pictures (Plates 2-2 to 2-4) were taken in sequence from one survey station and can be edge matched to create a 150° view of the Athabasca Valley.

M. P. Bridgland, 1915

J. Rhemtulla, 1997

Plate 2-2: Athabasca River Valley, view N.E. (Station 57 – No. 460)

Original showcases large alluvial fan with dense young deciduous growth (centre). Open coniferous forest with an occasional deciduous patch is evident elsewhere. Retake shows that the conifer component has increased significantly in the alluvial fan. Both the foreground and flanks of the Colin Range behind the river show a marked increase in tree density. Increased human activity is apparent in the borrow pit (centre), power line, and power generating station (bottom right).

M. P. Bridgland, 1915

J. Rhemtulla, 1997

Plate 2-3: Athabasca River Valley, view E. (Station 57 – No. 461)

Sparse tree cover in the foreground of the original is primarily coniferous with a younger cohort of aspen below it. There are a number of denser aspen stands on the other side of the river. Retake shows increased tree cover in foreground, along east side of the river, and lower flanks of the Colin Range. The aspen component has diminished greatly. The power generating plant is an example of increased human activity in the valley.

M. P. Bridgland, 1915

J. Rhemtulla, 1997

Plate 2-4: Athabasca River Valley, view E.S.E. (Station 57 – No. 462)

Historical photograph shows patchy vegetation throughout the valley. Sparse tree cover prevails. Dense young growth, likely coniferous, is evident behind dense coniferous stand centre-right. Coniferous species dominate, but deciduous trees are also apparent. Repeat photograph shows increased tree density throughout, especially in valley bottom. Deciduous component appears to have declined. Apparent homogeneity of vegetation overall on the landscape has increased .

M. P. Bridgland, 1915

J. Rhemtulla, 1997

Plate 2-5: Henry House Flats (Station 58 – No. 467)

Original photograph shows grassland in foreground with scattered bushes, likely buffalo berry (*Sheperdia canadensis*), and a fair amount of downed woody material. Grassland persists in retake, although shrub density has declined, and forest cover has increased. Tree cover has also increased markedly on the flanks of the Colin Range. Note that the two clusters of trees, centre left, are present in the original as young trees. (c.f. Plate 2-6 for a different perspective of the grassland complex.)

M. P. Bridgland, 1915

J. Rhemtulla, 1997

Plate 2-6: Colin Range and Henry House Flats (Station 38 – No. 308)

Forest encroachment on grassland is evident in the Henry House Flats, centre right (c.f. Plate 2-5). Greatly increased forest cover is also apparent on the east side of the Athabasca River and the flanks of the Colin Range above it. The two rail lines in the original photograph have today been merged into one; the dismantled line is now used as a local road. Both of these are visible in the retake (local road just discernible along right edge of photo 3 cm above river), as is the highway which now runs along the railroad. The burned area at the base of Hawk Mountain (bottom centre), resulted from a prescribed burn set by park managers in 1989.

M. P. Bridgland, 1915

J. Rhemtulla, 1997

Plate 2-7: Snaring River (Station 60 – No. 482)

The most obvious feature in the original is the retaining wall built along both sides of the Snaring River to prevent flood waters from washing out the bridge. Remnants of these great rock-filled timber cradles are still evident on the ground today. Bushes and deciduous trees along the north side of the river have been replaced by dense coniferous growth in the retake, likely due in part to the changes in water regime (c.f. Plate 2-8).

M. P. Bridgland, 1915

J. Rhemtulla, 1997

Plate 2-8: Snaring River confluence (Station 38 – No. 309)

The paired photographs show that the width of the Snaring River has declined, likely due in part to the retaining walls constructed in the 1910's to prevent flood damage to the bridges (c.f. Plate 2-7). The original photograph suggests that the lake (left-centre) is connected to the Snaring River in 1915. The lake outlet to the Snaring has diminished significantly in the repeat photograph. Differences in the wetland complex are hard to discern due to the scale of the photographs, and may be due in part to seasonal differences in water levels if the photographs were taken at different times of year (dates of Bridgland photographs are unavailable). Shrub cover on the wetland area adjacent to the lake may have declined in favour of greater herbaceous cover. Increased human activity is apparent in the large rectangular overflow campground (lower right) and the new highway (lowest of three transport corridors). Cracks in the original are due to the breaking of the glass plate negative.

M. P. Bridgland, 1915

J. Rhemtulla, 1997

Plate 2-9: Esplanade Wetland (Station 38 – No. 311)

Fragmentation of the wetland area by human infrastructure has increased. Differences in the wetland complex are hard to discern due to the scale of the photographs, and may be related in part to seasonal differences in water levels if the photographs were taken at different times of year (dates of Bridgland photographs are unavailable). Increased tree cover and density are visible in the left midground. The Athabasca River channel (bottom left) has cut further west and may soon hit the highway, and the two small channels flowing from it appear to have decreased water flow.

M. P. Bridgland, 1915

J. Rhemtulla, 1997

Plate 2-10: Study area – North end (Station 63 – No. 508)

The paired photographs give an overview of the north end of the study site. The Esplanade wetland (c.f. Plate 2-9) is in the centre midground. North-east of the wetland, the original photo shows an open grassland area with a significant deciduous tree component at the base of the mountains. Coniferous forest has encroached on this entire area in the retake, leaving behind only a number of smaller discontinuous grassland patches. Increase in forest cover and density is also apparent just above the wetland complex.

M. P. Bridgland, 1915

J. Rhemtulla, 1997

Plate 2-11: Study area – South end (Station 34 – No. 282)

The paired photographs provide an overview of the south end of the study site. Tree encroachment into the Henry House grassland (c.f. Plates 2-5 & 2-6) is apparent photo-centre. The small lake (centre-right) between the grassland complex and Snaring River (c.f. Plate 2-8) has an obvious outlet to the Athabasca River in 1915, which has been severed by the transport corridors in the matching photo. Increased tree cover and density is apparent on the east bank of the river (bottom centre-left). The dappled shadow effect due to clouds in both pictures makes detailed analysis difficult.

M. P. Bridgland, 1915

J. Rhemtulla, 1997

2.4.2 Quantitative analysis

The results of the quantitative analysis confirmed the qualitative observations. Open Forest, Forest(A) and Forest(B) cover types declined in area from 16% to 5%, 15% to 4%, and 16% to 10% respectively (Figure 2-2). Forest(C) and Forest(D) increased in photograph area from 3% and 0% in 1915 to 23% each in 1997 (Figure 2-2). Total forested area overall in the study site increased from 50% to 65%. Eighty percent of the area in forested cover in 1915 remained within that cover type in 1997, 10% of it was lost to anthropogenic activity, and the remainder to other vegetation types (Table 2-1). Within forested stands, 85% of Open Forest, 83% of Forest(A), 88% of Forest(B), and 83% of Forest(C) area in the 1915 photographs changed to forested vegetation types with greater crown closure in the 1997 photographs (Table 2-2). There was very little forested area where crown closure decreased between the two sets of photographs (Table 2-2). Young tree cover fell from 7% of photograph area in 1915 to almost 0% in 1997 (Figure 2-2) and eighty-six percent of this Young Tree growth in 1915 shifted to Forest categories in 1997 (Table 2-1).

Forest encroachment on grasslands was also apparent in the analysis. Only 25% of area under Herb cover in 1915 remained in that cover type in 1997, while 61% changed to one of the forested cover types (Table 2-1). Overall, herbaceous cover declined by 50% of photograph area between 1915 and 1997 (Figure 2-2). Sixty-four percent of shrub area in the 1915 photographs shifted to forested cover types (Table 2-1) while total photograph area in shrub cover declined from 13% to 4% (Figure 2-2). The Wetland cover type remained relatively stable over time. The overall photograph area remained constant at 2% (Figure 2-2). Just over 60% of it did not shift categories, 17% shifted to forested categories, and 17% was replaced by water (Table 2-1).

The Sand/Gravel category underwent some dramatic changes: 51% of the 1915 photograph area shifted to herb in 1997, 33% to forested categories, and 14% to water (Table 2-1). Finally, about 50% of anthropogenic cover in 1915 reverted to vegetated categories in 1997 (Table 2-1). However, total anthropogenic cover increased from 1% to 8% photograph area (Figure 2-2). The additional area seemed to have been appropriated from a number of different vegetation categories (Table 2-1).

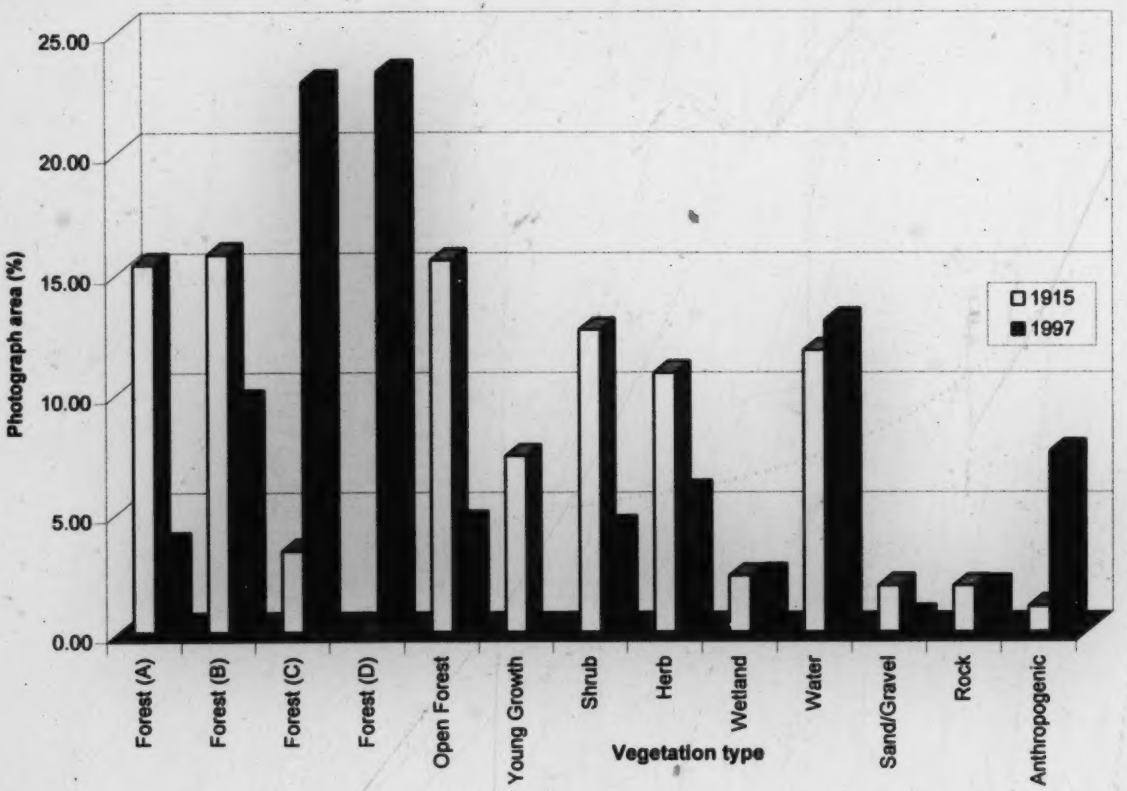

Figure 2-2: Vegetation Change, 1915 – 1997. Individual photographs were summed by cover type (in pixel units); overall change was calculated from summed data (in %). It is important to stress that these numbers do not represent absolute change within the study site. They are a measure of the absolute change of polygon area on the photographs, and provide only an estimate of the extent of change on the landscape. Forest crown closure classes are: A: 16-30% cover, B: 31-50%, C: 51-70%, D: 71-100%.

Table 2-1: Transition matrix showing vegetation changes from 1915 to 1997 (in %). Cross-tabulations for individual repeat pairs were first summed (in pixel units). For each 1915 vegetation type, percentage of that vegetation type changing to other types was calculated. Thus rows sum to 100% (e.g. of the total area that was Forest in 1915, 80.37% remained Forest in 1997, 2.20% moved to Open Forest, 0% moved to Young Tree, etc.). Shaded cells show area that remained stable between the two dates.

1915 \ 1997	Forest	Open Forest	Young Growth	Shrub	Herb	Wetland	Water	Sand / Gravel	Rock	Anthro-pogenic
Forest	80.37	2.20	-	3.06	0.94	0.77	1.86	0.06	0.13	10.61
Open Forest	64.45	11.52	-	2.90	4.69	0.12	5.14	-	-	11.59
Young Growth	85.91	4.44	5.64	3.17	1.51	0.37	1.42	-	0.18	3.02
Shrub	56.18	7.39	0.23	14.06	2.72	2.46	10.56	2.46	0.35	3.58
Herb	56.60	4.05	0.13	3.81	25.29	0.53	2.38	-	0.02	7.18
Wetland	15.47	1.18	-	2.39	1.66	62.63	16.77	-	0.01	-
Water	8.44	1.61	-	2.85	3.73	1.91	78.48	2.33	0.10	0.59
Sand/Gravel	24.52	8.14	-	2.02	51.06	-	13.79	0.47	-	-
Rock	4.08	1.08	-	0.23	-	-	-	2.26	92.35	-
Anthropogenic	34.81	0.24	-	-	11.74	-	-	-	-	53.21

Table 2-2: Transition matrix showing changes in crown closure within forested area from 1915 to 1997 (in %). (See caption, Table 2-1, for greater detail).

1915 \ 1997	Open Forest (6-15%)	Forest A (16-30%)	Forest B (31-50%)	Forest C (51-70%)	Forest D (71-100%)
Open Forest	14.69	3.57	11.51	26.71	43.53
Forest (A)	5.66	11.05	20.05	41.01	21.63
Forest (B)	0.33	1.31	9.96	41.81	46.60
Forest (C)	0.11	-	0.59	16.78	82.53
Forest (D)	-	-	-	-	-

A summary of major successional trends in the photographs can be seen in Figure 2-3. Water, rock, and wetland were the most stable cover types. The greatest amount of change was in favour of forested vegetation types, particularly Forest(C) and Forest(D), which have the greatest crown closure. Other trends included the shift from Sand/Gravel to Herb, and from Shrub, Wetland, and Sand/Gravel to Water.

2.5 Discussion and Conclusions

The results of both the qualitative and quantitative analyses of the repeat photographs suggest a number of striking changes in the montane landscape of Jasper National Park over the last century. There has been a dramatic increase in the extent and degree of crown closure of coniferous forest in the study area, accompanied by a decrease in the occurrence and extent of deciduous stands, grasslands, and young regrowth. The overall appearance of the valley bottom is less patchy and more homogeneous. Increases in anthropogenic activity have resulted in modifications in the hydrological regimes of the Snaring and Athabasca Rivers and in the Esplanade Wetland.

Similar changes have been reported throughout the mountains of the southern Canadian Rockies and the western United States. A widespread increase in density and extent of coniferous stands, forest encroachment on grasslands, and decline in aspen

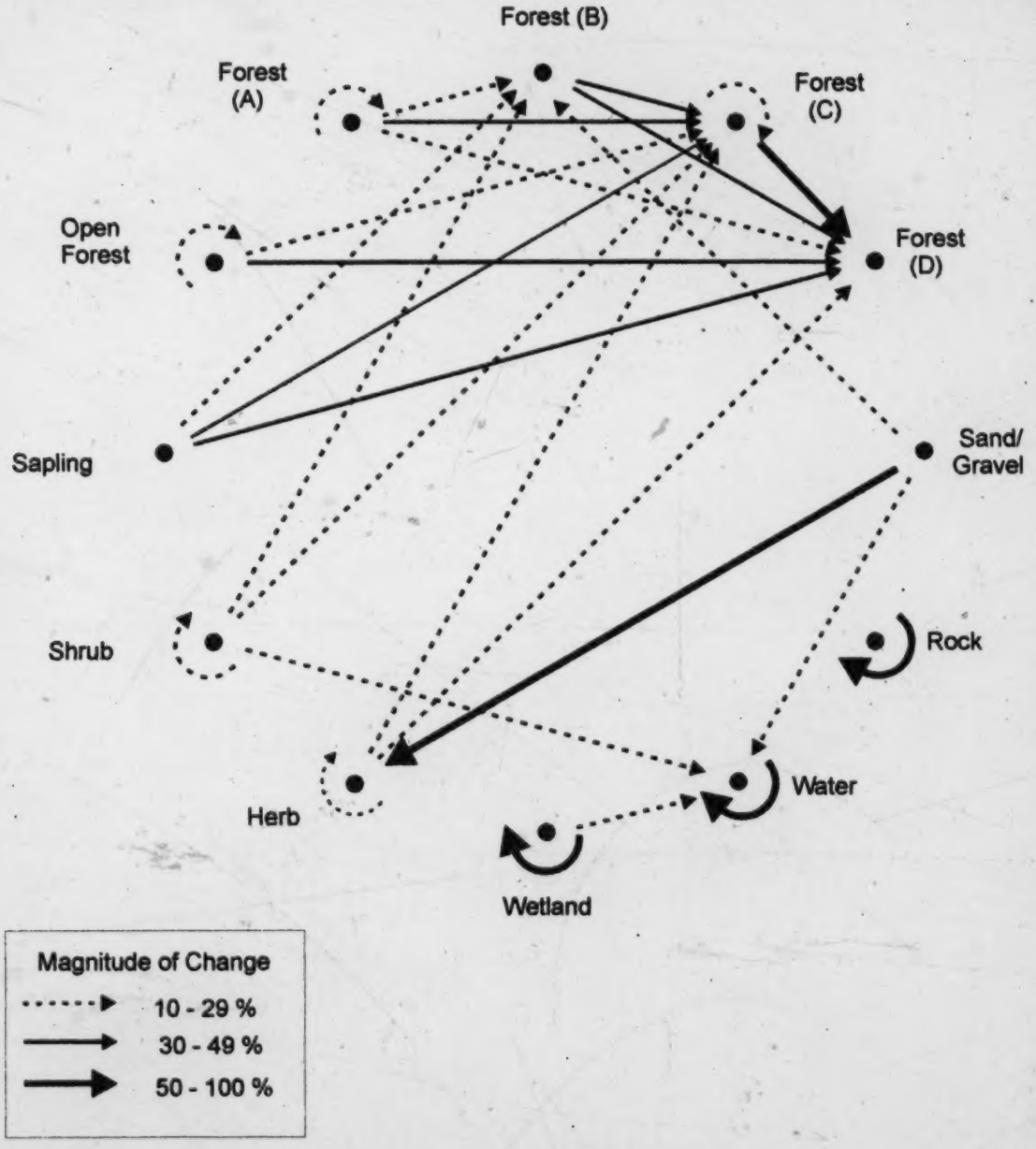

Figure 2-3: Major successional trends, 1915 - 1997. Arrows depict major directions of change between cover types in photograph area (%). Data are calculated as in Tables 2-1 and 2-2, with anthropogenic cover types removed from analysis to better reflect natural successional processes occurring in the study area. Forest crown closure classes are: Open Forest: 6-15%, Forest A:16-30%, B:31-50%, C:51-70%, D:71-100%. Diagram design after Hester *et al.* 1996.

stands and other early successional vegetation has been observed in Banff National Park (Achuff *et al.* 1996), Waterton Lakes National Park (Barrett 1996), Montana and Idaho (Gruell 1983), Colorado (Veblen and Lorenz 1986, Mast *et al.* 1997) and Yellowstone National Park (Houston 1982).

The reasons for these changes are hard to disentangle. The Bridgland photographs were taken in 1915, about 25 years after the widespread fires in 1889 burned much of the main valley of the park (Tande 1977). Fires in 1905 and 1908 were also recorded in parts of the study area (Tande 1977). The pictures, therefore, record an ecosystem recently affected by fire, and recovering from its effects. The repeat photographs, on the contrary, depict an ecosystem which has seen almost no fire in the intervening 80 years. In a sense, the photographs depict an ecosystem at the two extremes along a successional continuum – shortly after a major fire, and perhaps shortly before the next one.

Many researchers have suggested that area burned this century in the park is much lower than that recorded in the previous three centuries (Tande 1977, Van Wagner 1995). Reasons for the decline are debated but include increased fire suppression and prevention (see Chapter 1 for a more detailed discussion). The historical pictures may support the hypothesis of more frequent fire events in the past. Many of the open forest stands include scattered standing burned stems interspersed within living trees. This suggests a fairly open pre-burn stand, and/or low tree mortality due to the fire, both of which are consistent with frequent low-intensity fire regime. In contrast, the two fire events visible in the 1997 photographs, both the result of prescribed burns set in the last 10 years (D. Macdonald, JNP Warden, pers. com. 1998) appear as dense stands of standing burned timber. The pre-burn stands in these pictures were clearly heavily treed, and stand mortality and therefore fire intensity was high. Within the Bridgland photographs examined, there were no dense stands of burned timber. Although the visual appearance of ten-year prescribed burns may not be comparable to that of a twenty-five-year natural burn, it is probable that the forests in the area prior to the fires of the late 1800s and early 1900s were not the dense, closed canopy forests lining the valleys today in Jasper.[1]

[1] It is also possible that standing burned timber was removed by local inhabitants in the park, had already fallen, or was burned again in subsequent fires and thus no longer apparent on the landscape by 1915.

If a low-intensity, high-frequency fire regime was indeed characteristic of the montane ecoregion, as fire history studies suggest (Tande 1977), high fuel accumulation such as that visible today would probably have been uncommon. Frequent low-intensity ground fires would maintain open forests and patchy early successional vegetation, and prevent the kind of fuel accumulation visible in the valleys today. Decades of reduced fire frequency, on the other hand, may eventually result in a fundamental change in the fire regime to one of infrequent, high intensity fires, with much fuel accumulation in the intervening period (Heinselman 1975).

It is difficult to ascertain if the increased forest crown closure apparent in the photographs is the result of an increase in stem density, or simply an increase in the age and size of trees already occupying the site in the 1915 photographs. Lodgepole pine, the dominant species in much of the montane forests, typically seeds in during the first few years following a fire, thus it is probable that little new regeneration has taken place since the last fire, although other species, such as spruce, may have established in the understorey over time. Although it is hard to be sure, many of the trees in the 1915 photographs appear to be of the right size to have germinated following the 1889 fire. A more detailed study of the ages of these stands may shed more light on the dynamics of these forests over the last century.

There is also evidence to suggest that changes in climate might have contributed to shifts in vegetation. More mesic climatic conditions over the last century in the Front Ranges of Colorado may have helped tipped the competitive balance from drought-tolerant grasses to trees, thus facilitating forest encroachment on grasslands (Veblen and Lorenz 1991). In Yellowstone, a warmer and drier climate this century has contributed to enhanced hydrosere succession, leading to a decline in riparian shrubs and seral ponds, and invasion of trees on former braided stream channels (Houston 1982).

Long-term climate patterns in Jasper and their ecological effects are less straight-forward. Reviews of glacier movements in the Canadian Rockies suggest that glacial advances during the Little Ice Age reached their maximum extents during the early 18th and mid-late 19th century (Luckman 1986), presumably coinciding with cooler temperatures. During the past century, glacial retreat was very rapid from 1920 to 1950, and then tapered off over the following three decades (Osborn and Luckman 1988). The

instrumental climate record suggests a roughly analogous scenario: an increase in temperatures between 1920 and the early 1940's, followed by a sharp decrease in the 1950's, brief recovery in the 1960's, and subsequent decline into the 1980's (Luckman 1990). The ecological effects of changing climate are not clear. Several studies have found that the cooler climate associated with the Little Ice Age corresponded with a decrease in fire frequency during that time in the Canadian Rockies (Johnson and Larsen 1991, Masters 1990); more recent research in the boreal forest of eastern Canada suggests that fire frequency was higher during the Little Ice Age than during the warmer period which followed it (Bergeron and Archambault 1993). Although it is hard to decipher the precise connections between climate, vegetation, and fire regime, it is possible that the shift in climate has contributed to the observed changes in vegetation, both directly by favouring certain plant species and indirectly by contributing to a decrease in fire frequency.

The role of herbivory in shaping vegetation patterns is also a matter of discussion. Increased grazing pressure by elk on grasslands in Yellowstone reduced competition from grasses and increased patches of bare mineral soil, thus facilitating invasion by tree species (Houston 1982). Browsing of aspen trees has also been cited as a major cause for declining health of aspen stands in Banff National Park (Achuff *et al.* 1996). In Jasper, it appears that increasing elk populations have affected the regeneration of aspen stands (P. Achuff, pers. comm.). It seems unlikely that elk pressure would be sufficient to explain the decline in aspen and grasslands over the last century in Jasper, but it has likely contributed to these processes.

Finally, direct human activities on the landscape have affected vegetation patterns and processes. In addition to our intervention in fire processes in the park, the construction of transportation and utility corridors, and various other infrastructure has either directly replaced existing natural vegetation, or affected the processes which shape it. The hydrological regime in the study area has been impacted by several of these human activities. Retaining walls along the Snaring River probably reduced flooding events, and the apparent impoundment of the adjacent lake may have led to rising water levels and a replacement of the shrub vegetation by wetland species. Fragmentation of the

Esplanade Wetland by transport corridors has probably affected nutrient and sediment flow through the area.

In all likelihood, a combination of several factors has led to the vegetation changes apparent in the Athabasca Valley over the last eighty years. The effects and implications of these changes on biodiversity, and on wildlife habitat availability and quality, is the issue which currently faces park management. At the core is whether both of these states are within the natural range of variation expected for this ecosystem, whether the current state of the park is a cause for concern, and whether the present landscape can support whatever values park management deems important now and into the future. I will address the overall implications of these changes, and possible responses to them in Chapter 4.

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CHAPTER 3

A View From Above: Mapping vegetation change through aerial photographs

3.0 Introduction

In Chapter 2, I used a set of repeat oblique historical photographs to determine how the vegetation in the montane ecoregion of Jasper National Park has changed from 1915 to 1997. In this chapter, I use standard methods of aerial photograph interpretation to analyse vegetation changes in greater detail for the period between 1949 and 1991. Although the time period studied is shorter, the use of aerial photographs permits a more detailed interpretation of vegetation attributes, greater resolution of features, absolute measures of change, and spatial analysis of the resulting maps.

Aerial photographs have been used in many studies to analyze changes in landscape features over time. Early work included qualitative visual interpretation of photographs and simple quantitative analyses of cover classes, akin to much of the current research being conducted with oblique photographs (Barnes 1989, Foster 1992, Liegel 1988). With the advent of Geographic Information Systems (G.I.S.), more complex spatial analyses and modeling are now possible.

Studies of landscape change using G.I.S. have focused generally on two major lines of inquiry. The first has been the changing landuse and landcover patterns in areas of intensive human activity. Research from places as widespread as Scotland (Hester *et al.* 1996), Thailand (Fox *et al.* 1995), Quebec (Jean and Bouchard 1991), and coastal British Columbia (Boyle *et al.* 1997), all suggest increasing human landuse on the landscape, a consequent decrease in total 'natural' area, and structural and compositional changes in the vegetation in these remaining natural areas. Other researchers have focused on vegetation changes in protected areas – areas supposedly shielded from the effects of human activity. Studies in a prairie reserve in Kansas (Knight *et al.* 1994),

national forests in Oregon and Washington (Lehmkuhl *et al.* 1994), the Front Ranges of Colorado (Mast *et al.* 1997), the interior forests of British Columbia (Taylor and Hawkes 1997), and along the St. Lawrence River in Quebec (Jean and Bouchard 1991) have all demonstrated that significant changes in vegetation have occurred in the past fifty years, many of them due to indirect human activities. For example, deviation from the historical fire regime is cited as a major contributor to many of the changes described, from gallery forest encroachment into surrounding tall-grass prairie (Knight *et al.* 1994), to increased canopy density in coniferous forests (Lehmkuhl *et al.* 1994, Mast *et al.* 1997, Taylor and Hawkes 1997) and the encroachment of shrublands in wetlands (Jean and Bouchard 1991).

Descriptions of specific changes that have occurred in ecosystems similar to the montane ecoregion in Jasper have been detailed elsewhere in this thesis and will not be reiterated here (see Chapters 1 and 2). In this chapter, I discuss how vegetation patterns in general, and forest stand structure and composition in particular, have changed from 1949 to 1991. Specific hypotheses addressed are detailed in Chapter 1.

3.1 Study Site

Jasper National Park (52 N, 118 W) is located in the Rocky Mountains of Alberta, about 400 km west of the city of Edmonton, and occupies a total of 10 880 km^2 (Holland and Coen 1982b). The study site was located in the lower Athabasca River valley, in the central region of the park, north-east of Jasper townsite (see Figure 2-1). It lay within the montane ecoregion, which occupies the lowest park elevations (±1000 to ±1350m asl). The study area extended from Jacques Creek and Vine Creek in the north to the confluence of the Maligne and Athabasca Rivers in the south. Boundaries of the study site along the east and west sides of the valley corresponded to the upper limits of the montane ecoregion, where there was adequate airphoto coverage. In a few areas, the 1991 airphoto coverage did not extend to the limits of the montane boundary, and the study area was thus reduced somewhat. The study area was approximately 14 km long by 4 km wide, and occupied a total of 64 km^2, 8.5% of the total montane ecoregion within the park.

The macroclimate of the park falls into the transitional zone between the Cordilleran and Continental climate regions of Canada (Seel and Strachan 1987). Regional climate is difficult to characterise because of both the strong gradients in physiographic influences that shape micro-climate, and the relative lack of comprehensive data. The climate of the montane ecoregion is generally the warmest and driest in the park, and has been classified as a Dfc climate (cold, snowy forest climate with no distinct dry season and cool short summers) (Holland and Coen 1982a). Mean daily minimum and maximum temperatures are 7.3°C and 23.1°C in July, and –16.7 °C and –6.1°C in January, the hottest and coldest months respectively, as measured at Jasper townsite (Stringer and LaRoi 1970). Average annual precipitation in the montane is 471 mm (averaged throughout the montane ecoregion of the park) with slightly more precipitation falling during the summer rather than winter months (Holland and Coen 1982a).

The study area lies chiefly within the Front Ranges of the Canadian Rockies; the peaks are underlain by Late Palaeozoic limestone, and the valleys by Mesozoic shale (Gadd 1986). Surface material in the Athabasca River valley is primarily till and glaciofluvial deposits composed of sandstone and quartzites with some slate and limestone (Stringer and LaRoi 1970).

3.2 Methods

3.2.1 Airphoto Interpretation

Airphoto coverage of the study site at two dates, 1949 and 1991, was selected for interpretation. The 1949 aerial photographs (1:40000, Super XX B/W film, Rolls AS 143-145, flown September 15) were the first to be flown in the park. The 1991 photographs (1:20000, Pan XX B/W film, Roll AS4212, flown September 25) were the most recent medium-scale series available. Working copies of the 1949 photographs were enlarged to 1:20000, so that the minimum mappable unit would be equivalent for both sets.

A hierarchical land cover classification system was developed based on the Alberta Vegetation Inventory widely in use in the province of Alberta (Nesby 1997), and the Ecological Land Classification (E.L.C.) used in Jasper National Park (Holland and Coen 1982a). Attributes included cover type; stand structure, species composition and crown closure for forested polygons; and several stand modifiers (windfall, burn, and snags) (see Appendix 3).

Air photos were covered with transparent acetate overlays. Areas of homogenous vegetation cover (compositionally and structurally) were delineated with the aid of an Abrams 2-4 stereoscope (model CB-1). Minimum polygon size was approximately 0.5 cm^2 (2 ha). Linear attributes (e.g. roads, streams) over 30m (1.5 mm) wide were delineated as separate polygons. The 1991 and 1949 airphotos were interpreted as independently of one another as possible, in an attempt to minimise interpreter bias.

3.2.2 Ground-truthing

Ground-truthing to check the accuracy of the interpretation was conducted on a subset of the polygons. Ten transects varying in length from 1.8 to 3.2 km were traversed in different areas of the study site. At equal distances along the transect (usually 100-150 m apart), vegetative characteristics were recorded, including vegetation type, dominant overstorey and understorey tree species, soil moisture characteristics, and E.L.C. vegetation type. Four crown closure measures were taken at each site (facing outwards at the corners of a 10m x 10m plot) using a spherical densiometer. Crown closure measures were averaged for each site. At sites with mixed canopy composition, diameter at breast height (DBH) of all trees within a $100m^2$ circular plot was measured. DBH data were used to calculate percent composition of individual canopy species.

A total of 234 ground-truthing plots were sampled. Similar field data from a further 61 plots were obtained from a concurrent study (M. Norton, unpublished data). Plot data were used to confirm interpretations of 132 polygons (some polygons had multiple plots) in the 1991 vegetation map (approximately 16% of the total polygons). Because ground-truthing data were used to both aid the initial interpretation process and evaluate its accuracy in an iterative fashion, it was not possible to use the data to estimate the overall accuracy of the final 1991 vegetation map. The collection of further data for

this purpose was not feasible due to lack of time. Ground-truthing of the 1949 photographs was, obviously, not possible. Following the initial interpretation stage, however, a sub-set of 1949 polygons were compared with the corresponding area on the 1991 vegetation map to ensure accuracy and consistency of the interpretation. Some informal field checking of a few polygons which were hard to interpret was carried out throughout the process.

3.2.3 Map creation

Once the interpretation was complete, polygons were transferred to transparent sheets of acetate overlaid onto an orthophoto base map (1:20000). Thirty ground control points throughout the study area were collected with a Global Positioning System unit. The final geocorrected composite acetates were digitized, edited and labeled using ArcInfo software. Attribute data for the polygons were entered into a database using Microsoft Access 2.0 software. The coverages and databases were subsequently imported into the IDRISI G.I.S. software where they were linked for further analysis (Eastman 1997). Vector files were transformed to raster coverages with a cell resolution of 10m. The quality of this transformation was deemed adequate by both visual inspection, and by comparing patch statistics for the vector and corresponding raster layers.

3.2.4 Spatial and Statistical Interpretation

Analysis of the 1949 and 1991 coverages included only the area of common overlap between the two maps. Descriptive statistics were calculated by cover type, for the entire study area, and by overstorey species composition, for forested areas. Coniferous forest stands, which dominated the forested area, were further analyzed by stand structure and crown closure. In each case, a coverage of the particular attribute was created. Overall percent area within each category of the attribute was calculated in IDRISI. Patch, class, and landscape metrics were then calculated using the program Fragstats (McGarigal and Marks 1995).

Patch level data were imported into SPSS for statistical analysis (Norušis/SPSS Inc. 1993). Patch size data were log transformed and tested for normality (Sokal and Rohlf 1981, Zar 1974). Changes in average patch size (in log area) were analysed

between years with simple factorial analysis of variance (ANOVA) and within cover type (by year) with Student's t-tests (Sokal and Rohlf 1981, Zar 1974).

Once separate analysis of the coverages was complete, a number of overlay analyses were carried out in IDRISI. Image cross-tabulation between 1949 and 1991 was executed by cover type and crown closure (see Chapter 2 for further details on this procedure). Two phenomenon were examined in greater detail: forest encroachment and changes in crown closure. An aspect map was derived from an existing digital elevation model of the study area. The continuously varying surface was reclassified into 5 categories: flat (no slope), north (316°-45°), east (46°-135°), south (136°-225°), and west (226°-315°). A map of distance to nearest forest pixel was created using the DISTANCE module (Eastman 1997). Distance was reclassified into 6 classes: 0-99m, 100-199m, 200-299m, 300-399m, 400-499m, and 500-599m. Finally, a map showing natural disturbance events (fire and windfall) was also created.

Residual (binary) maps showing areas of forest encroachment (areas of grass cover in 1949 that changed to forest cover in 1991), grass cover in 1949, and the total study area, were created. Each was overlaid in turn with the aspect and the distance to nearest forest patch maps. Percentage area occupied by aspect and forest proximity classes was calculated.

Changes in forest crown closure between 1949 and 1991 were lumped into 3 categories: overall increase, overall decrease, and no apparent change. Residual maps for each of the categories were produced, as were maps of total forest cover in 1949, and total study area. Each coverage was overlaid with aspect, distance to nearest forest patch, and disturbance maps. Percentage area occupied by aspect class and proximity to nearest forest patch was calculated.

3.3 Results

The distribution of cover types in the study area suggests an apparent increase in homogeneity across the landscape as a whole from 1949 to 1991 (Figure 3-1). Forested area increased from 3716 to 4171 ha; smaller increases were apparent in the shrub (226 to 358 ha) and anthropogenic (144 to 248 ha) categories (Figure 3-2). Young tree growth disappeared completely from the landscape, and decreases in forb (234 to 69 ha),

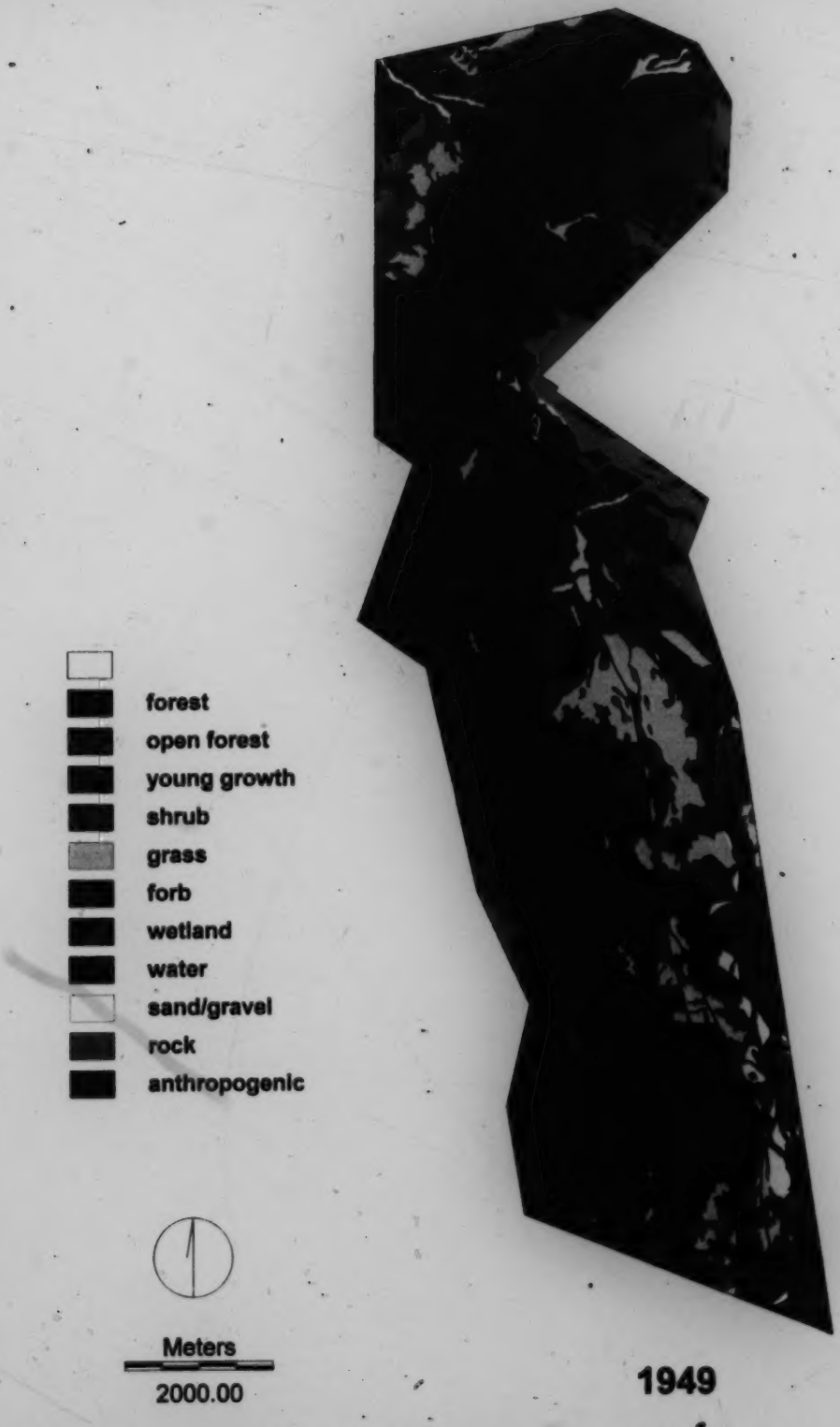

Figure 3-1: Distribution of cover types in the study area, 1949 and 1991

1949

1991

area, 1949 and 1991

grass (339 to 215 ha), and rock (228 to 167 ha) cover were also observed (Figure 3-2). Area occupied by open forest, wetland, water, and sand/gravel was about the same at the two dates (Figure 3-2).

The total number of patches on the landscape fell from 476 to 407 (Table 3-1). Decline in patch number was most evident in forest (108 to 75), young tree growth (15 to 0), forb (53 to 20) and sand/gravel (35 to 20) cover types (Table 3-1). Increase in patch number was observed for shrub (42 to 55), rock (32 to 44), and anthropogenic (9 to 15) categories (Table 3-1). Mean patch size did not differ significantly for any cover type, although variability in patch size was extremely high (Table 3-1) (see Appendix 4 for details of statistical tests).

Within the forested region of the study site, overstorey canopy species composition was similar (by proportional area) at the two dates, and coniferous stands were by far the dominant stand type on the landscape (>90% at both dates) (Figure 3-3). Total number of patches increased slightly (150 to 165), with the greatest increases occurring within the three mixed-wood canopy types (Table 3-2). Number of patches declined within the coniferous stand type (Table 3-2). Patch size distributions did not differ significantly by canopy composition (Appendix 4), although mean patch size increased for coniferous stands, and decreased for most other stand types (Table 3-2).

Within coniferous stands, single canopy stands dominated the forested area in both 1949 and 1991; proportional area by stand structure type remained roughly constant between the two dates (Figure 3-4). Total number of patches decreased overall (126 to 110), with a decrease in single canopy stands (116 to 81) and an increase in multiple canopy stands (10 to 27) (Table 3-3). Patch size distributions did not differ significantly for any of the stand structure types (Appendix 4), although mean patch size increased in single canopy stands and decreased in multiple canopy stands (Table 3-3).

There was a general shift toward increased crown closure within coniferous stands from 1949 to 1991 (Figure 3-5). The proportional area of open forest remained about equal at both dates (approx. 3%), Forest(A) increased from 7 to 16%, Forest(B) and (C) declined from 22 to 14% and 45 to 36% respectively, and Forest(D) increased from 22 to 32% cover (Figure 3-6). Total number of patches declined from 362 to 306; the largest individual declines were observed in stands of (B) and (C) cover (Table 3-4).

Table 3-1: Patch number, mean patch size (ha) and standard error in 1949 and 1991 by cover type. Details of statistical analysis are in Appendix 4.

Cover Type	Number of Patches		Mean Patch Size (ha)		Standard Error	
	1949	1991	1949	1991	1949	1991
Forest	108	75	34.41	55.61	16.10	24.97
Open forest	40	37	3.11	3.37	0.67	0.54
Young Growth	15	0	18.93	-	4.95	-
Shrub	42	55	5.39	6.51	1.13	1.28
Grass	57	54	5.94	3.98	1.71	1.39
Forb	53	20	4.41	3.44	1.06	0.79
Wetland	32	40	7.13	5.43	3.10	1.00
Water	53	47	15.39	17.00	10.88	10.50
Sand / Gravel	35	20	3.16	4.10	0.51	1.00
Rock	32	44	7.14	3.80	3.43	1.06
Anthropogenic	9	15	16.02	16.52	9.36	6.45
TOTAL	476	407	13.55	15.85	3.90	4.84

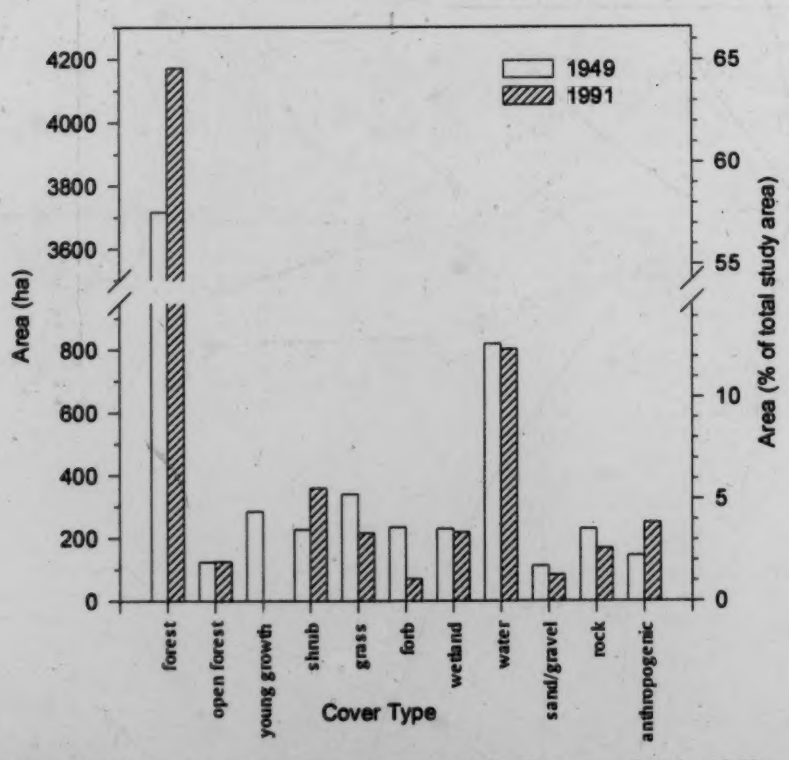

Figure 3-2: Landscape cover in percent area and absolute area (ha) in 1949 and 1991 by cover class.

Table 3-2: Patch number, mean patch size (ha) and standard error in 1949 and 1991 by overstorey species composition. C – coniferous, CD – coniferous dominant with deciduous component, M – mixed coniferous/deciduous, DC – deciduous dominant with coniferous component, D – deciduous. Details of statistical analysis are in Appendix 4.

Spp. Comp	Number of Patches		Mean Patch Size (ha)		Standard Error	
	1949	1991	1949	1991	1949	1991
C	107	79	32.65	48.51	15.24	21.76
CD	8	16	8.10	4.71	2.82	1.12
M	10	22	3.30	3.02	1.31	0.73
DC	6	24	2.29	4.75	0.35	2.23
D	19	24	5.86	3.42	1.96	0.88
TOTAL	150	165	24.77	25.28	10.91	10.53

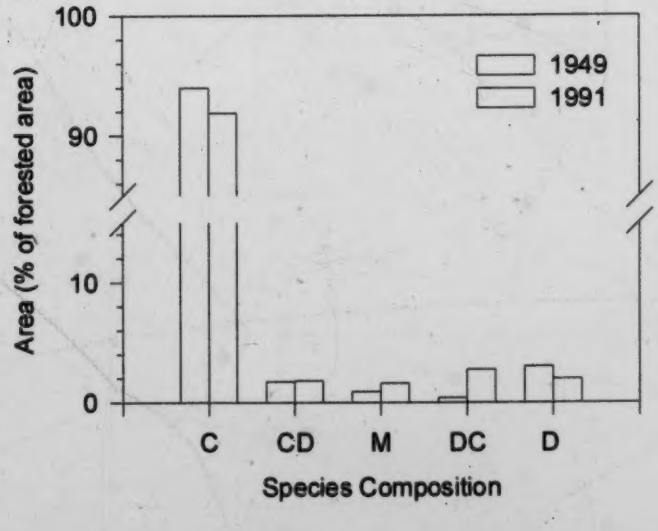

Figure 3-3: Overstorey species composition in 1949 and 1991 (expressed as % of forested area). C – coniferous, CD – coniferous dominant with deciduous component, M – mixed coniferous/deciduous, DC – deciduous dominant with coniferous component, D – deciduous.

Table 3-3: Patch number, mean patch size (ha) and standard error in 1949 and 1991 within coniferous forest, by stand structure type. Details of statistical analysis are in Appendix 4.

Stand Structure	Number of Patches		Mean Patch Size (ha)		Standard Error	
	1949	1991	1949	1991	1949	1991
Single	116	81	26.82	39.81	13.33	17.43
Multiple	10	27	29.22	12.65	15.69	3.68
Complex	0	2	-	23.76	-	5.22
TOTAL	126	110	27.01	32.85	12.32	12.89

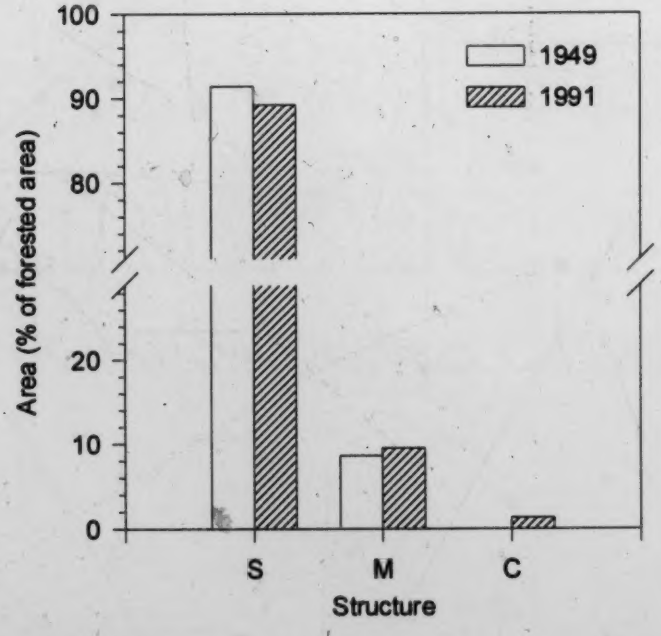

Figure 3-4: Stand structure within coniferous forest in 1949 and 1991 (expressed as % of forested area). S – single canopy, M – multiple canopy, C – complex canopy structure.

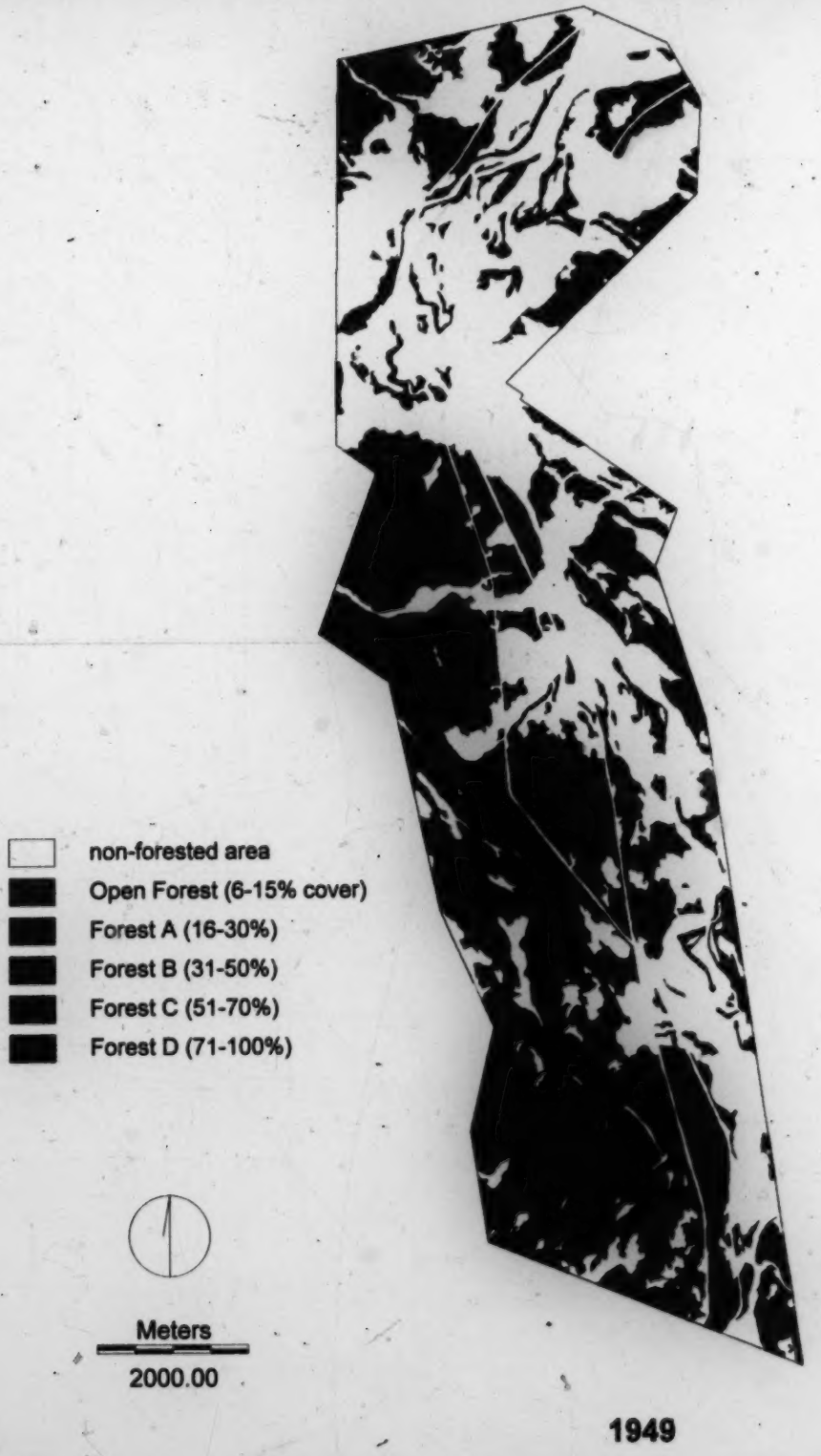

Figure 3-5: Crown closure in coniferous forests, 1949 and 1991

1949

1991

9 and 1991

Table 3-4: Patch number, mean patch size (ha), and standard error within coniferous forest, by crown closure class. Open Forest – 6-15% cover, Forest(A) – 16-30%, Forest(B) – 31-50%, Forest(C) – 51-70%, Forest(D) – 71-100%. Asterisk indicates a significant difference between 1949 and 1991 conditions at the 0.05 level; details of statistical analysis are in Appendix 4.

Density	Number of Patches		Mean Patch Size (ha)		Standard Error	
	1949	1991	1949	1991	1949	1991
Open Forest	40	35	3.11	3.00	0.67	0.46
Forest (A)	58	55	4.53*	11.61*	0.82	3.18
Forest (B)	112	90	7.19	5.77	1.42	0.65
Forest (C)	99	77	16.41	18.24	3.95	4.14
Forest (D)	53	49	15.09	25.92	5.60	9.67
TOTAL	362	306	9.99	12.87	1.45	2.00

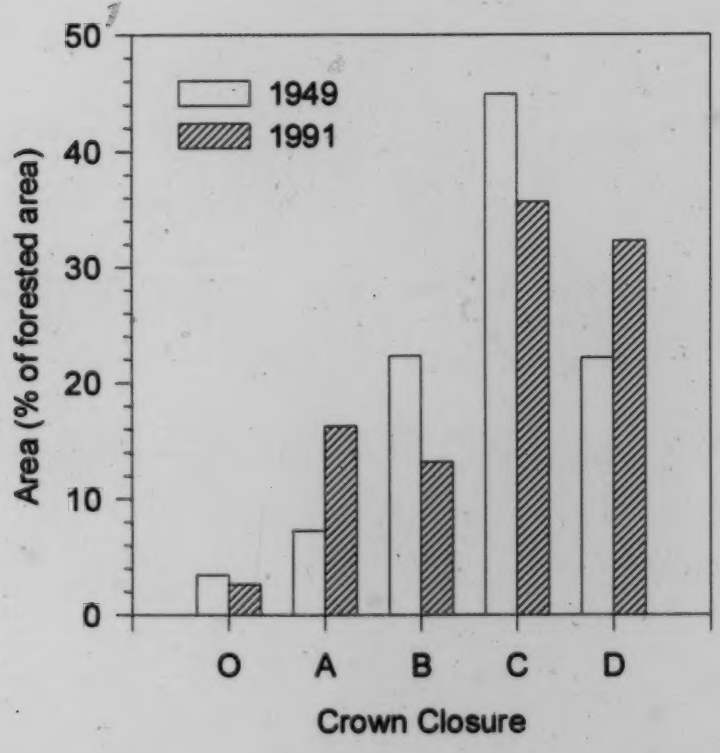

Figure 3-6: Change in crown closure, 1949 to 1991, within coniferous forest (expressed as % of forested area). O – 6-15% cover, A – 16-30%, B – 31-50%, C – 51-70%, D – 71-100%.

Within individual cover classes, there was a significant increase in patch size of Forest(A) stands ($t = -2.765$, $df = 111$, $p=0.007$) (Appendix 4, Table 3-4).

Transitions between cover types from 1949 to 1991 are shown in Table 3-5 and Figure 3-7. The greatest change was a general successional trend toward forest from most of the other cover types. Ninety percent of young growth, 64% of open forest, 41% of shrub, 34% of rock, 25% of grass, and 20% of wetland changed to forested stands over the 40 year period (Table 3-5). The most stable cover types were forest, water, and rock (Figure 3-7). Several cover types exhibited multiple transitional pathways. Less than 10% of forb cover in 1949 remained within that cover type in 1991, 51% changed to shrub, 15% to forest, and 10% to water (Figure 3-7). Forty-seven percent of wetland areas remained as such, while 20%, 15%, and 13% shifted to forest, shrub, and water, respectively (Figure 3-7). Finally, 23% of sand/gravel areas persisted in 1991, while 26% shifted to water, 17% to forb, 13% to wetland, 12% to forest, and 7% to shrub (Figure 3-7)..

Within the forested region of the study area, 97% of coniferous stands in 1949 continued to be dominated by coniferous species in 1991 (Table 3-6). The majority of stands dominated by either coniferous species with a small deciduous component, or of mixed composition in 1949, were dominated by coniferous species by 1991 (Table 3-6). Stands composed of primarily deciduous species with a small conifer component in 1949 were dominated by either deciduous or coniferous species in 1991 (Table 3-6). Finally, 49% of deciduous stands in 1949 were still classified this way in 1991, while 41% had become dominated by coniferous species (Table 3-6).

There was a general shift towards greater crown closure from 1949 to 1991 (Table 3-7, Figure 3-7). Increased canopy cover was observed in approximately 72% of the open forest, 61% of Forest(A), 59% of Forest(B), and 33% of Forest(C) (Table 3-7). Decreases in forest crown closure were observed in 17% of Forest(B), 22% of Forest(C), and 24% of Forest(D) (Table 3-7).

Areas of forest encroachment into previous grasslands are shown in Figure 3-8. Forest encroachment occurred with less frequency in flat areas, and greater frequency on south-facing slopes and within 100m of existing forest patches, as compared to the overall distribution of grass areas in 1949 (Table 3-8). Correlations between changes in

Table 3 - 5: Transition matrix showing vegetation changes from 1949 to 1991 (%). Cells show percentage of the 1949 cover type changing to 1991 types. Thus rows sum to 100% (e.g. of the total area that was Forest in 1949, 91.94% remained Forest in 1991, 0.37% moved to Open Forest, 0% to Young growth, etc.). Shaded cells show area that remained stable between the two dates. Kappa Index of Similarity varies between 0 (no similarity) and 1 (identical).

1991 / 1949	Forest	Open forest	Young growth	Shrub	Grass	Forb	Wetland	Water	Sand / Gravel	Rock	Anthro	Total Area (ha)	kappa index
Forest		0.37	-	1.37	0.42	0.02	0.51	1.64	0.32	0.78	2.63	3716.15	0.772
Open forest	64.38		-	1.46	4.30	0.61	-	0.67	0.42	0.35	4.21	124.52	0.221
Young growth	89.45	4.44		1.89	0.85	0.12	0.17	2.30	0.43	0.20	0.14	283.90	0
Shrub	40.66	2.04	-		0.03	0.55	8.37	5.53	0.65	0.24	1.76	226.21	0.3666
Grass	24.90	12.35	-	3.53		-	0.18	0.53	0.04	0.03	4.33	338.67	0.5253
Forb	15.17	4.78	-	50.73	0.06		4.80	10.48	0.99	0.02	4.47	233.90	0.0751
Wetland	20.42	0.69	-	15.06	0.07	0.95		12.93	0.75	0.02	2.13	228.02	0.4512
Water	5.53	0.30	-	3.77	0.04	2.99	5.43		4.36	-	0.92	815.92	0.7336
Sand / Gravel	12.29	0.36	-	7.04	0.05	16.63	13.48	25.91		0.04	1.44	110.73	0.2176
Rock	33.97	2.66	-	-	0.88	0.09	-	2.48	-		0.12	228.38	0.5873
Anthro	17.39	0.53	-	4.10	3.95	0.38	0.59	1.86	1.15	-		144.19	0.6885
Total Area (ha)	4170.53	124.72	-	358.26	214.90	68.87	217.40	798.92	81.94	167.30	247.75	6450.59	
kappa index	0.5736	0.2207		0.2266	0.8446	0.2619	0.474	0.7515	0.2954	0.8096	0.3941		

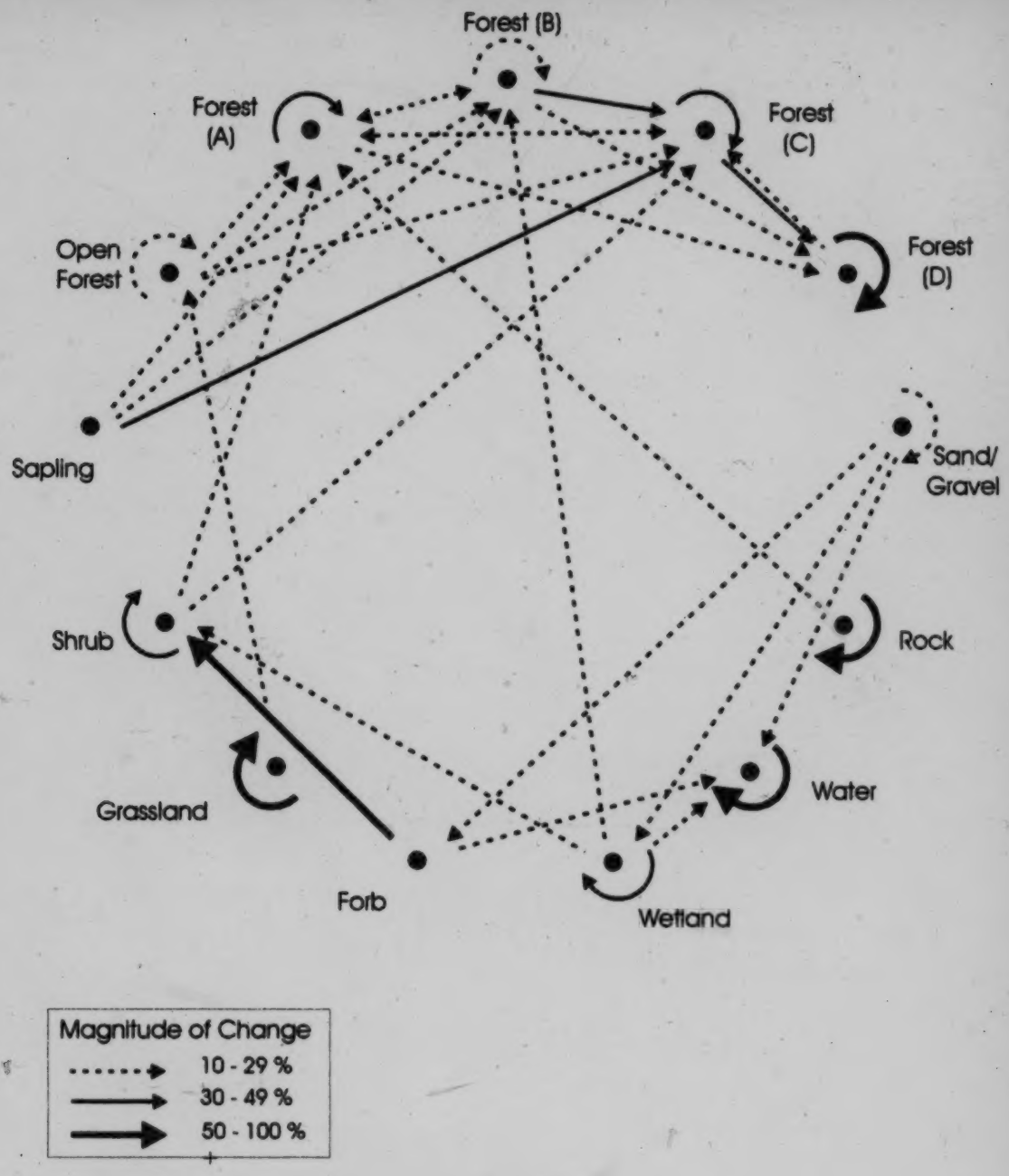

Figure 3-7: Major successional trends, 1949-1991. Arrows depict major directions of change between cover types (% area). Data are calculated as in Tables 3-5 and 3-6, with anthropogenic cover types removed from the analysis to better reflect natural successional processes occurring in the study area. Forest crown closure classes are: Open Forest: 6-15%, Forest A:16-30%, B:31-50%, C:51-70%, D:71-100%. Diagram design after Hester *et al.* 1996.

Table 3 - 6: Transition matrix showing changes in overstorey species composition from 1949 to 1991 (%). (See caption, Table 3-5 for greater detail.)

1949 \ 1991	C	CD	M	DC	D	Area (ha)	Kappa
C	[illegible]	1.48	0.85	0.12	0.53	3179.20	0.496
CD	86.13	[illegible]	3.66	1.21	0.76	59.33	0.065
M	64.16	8.07	[illegible]	2.14	24.33	46.74	0.004
DC	33.38	4.52	4.06	[illegible]	51.36	69.63	0.063
D	41.38	6.12	0.00	3.33	[illegible]	61.79	0.477
Area	3214.24	62.51	32.67	12.38	94.89	3416.69	
Kappa	0.419	0.062	0.005	0.363	0.308		[illegible]

Table 3 - 7: Transition matrix showing changes in coniferous forest crown closure from 1949 to 1991 (%). (See caption, Table 3-5 for greater detail.)

1949 \ 1991	Open forest (6 - 15%)	Forest A (16 - 30%)	Forest B (31 - 50%)	Forest C (51 - 70%)	Forest D (71 - 100%)	Total Area (ha)	Kappa Index
Open forest	[illegible]	13.57	28.92	24.18	5.59	106.02	0.27
Forest A	1.49	[illegible]	17.23	32.44	11.57	209.65	0.27
Forest B	0.64	16.06	[illegible]	41.86	16.93	727.42	0.15
Forest C	0.30	14.71	6.92	[illegible]	33.18	1406.74	0.13
Forest D	0.20	2.66	2.25	18.86	[illegible]	753.98	0.62
Total Area (ha)	42.85	436.45	359.37	1171.87	1193.27	3203.81	
Kappa Index	0.68	0.12	0.35	0.18	0.32		[illegible]

Figure 3-8: Forest Encroachment onto Grasslands, 1949 to 1991. Yellow represents areas of grassland cover at both dates; green represents areas of forest cover at both dates. Orange represents forest encroachment: areas of grass cover in 1949 which shifted to forest cover in 1991. Two types of encroachment are evident: the complete loss of some patches of grassland, and the decrease in size of others. Note that although the overall magnitude of change may seem small, concurrent increases in crown closure of forest stands (c.f. Figure 3-5, Figure 3-6, Table 3-7) have resulted in a major decline in open rangelands suitable for forage.

Table 3 -8: Area (in percentage) characterized by (a) forest encroachment (1949 grass -- > 1991 forest), and (b) total grass cover in 1949, by aspect and proximity to nearest forest cover. Columns sum to 100%. (e.g. 27.86% of grass cover in 1949 occurred in flat areas, 10.89% on north aspects, 26.16% on east aspects, etc.).

		Area (%)	
		Forest encroach-ment	Grass Cover 1949
Aspect	No slope	19.07	27.86
	North	9.08	10.89
	East	26.7	26.16
	South	29.6	21.34
	West	15.54	13.76
Proximity to nearest forest (m)	0 - 99	82.69	78.23
	100 - 199	12.56	17.69
	200 - 299	4.1	3.79
	300 - 399	0.66	0.28
	400 - 499	-	-
	500 - 599	-	-

forest crown closure and aspect and elevation were less conclusive (Table 3-9). Areas of unchanged crown cover were least likely to occur on west-facing slopes, and were evenly distributed elsewhere (Table 3-9). Decreases in crown cover occurred most frequently on south and west facing slopes while crown closure increases were slightly more prevalent on east, south, and west aspects (Table 3-9). Areas of unchanged or increased crown cover were most likely to occur at elevations below 1050m, while decreases in crown closure were most prevalent at middle elevations (Table 3-9). Crown closure increases were also more likely at the upper limit of the montane ecoregion (Table 3-9). Finally, decreases in crown cover were strongly correlated with both fire and windfall events

Table 3 - 9: Area (in percentage) characterized by (a) unchanged, (b) decreased, and (c) increased crown closure (1949 to 1991), and (d) total forested area (1949), by aspect, elevation, and disturbance event. Crown closure rows sum to 100% (e.g. In forested areas on flat land, 52.43% of forests did not change crown closure classes, 15.76% decreased in crown closure, and 31.82% increased in crown closure).

		Crown closure (% area)			Total Forest (ha)
		unchanged	decreased	increased	
Aspect	Flat	52.43	15.76	31.82	415.03
	North	51.69	16.64	31.66	519.03
	East	48.40	15.77	35.83	1189.49
	South	43.74	21.56	34.70	527.42
	West	35.85	29.10	35.06	552.84
Elevation (m)	< 1000	63.40	15.74	20.86	164.02
	1000 - 1050	43.94	16.83	39.23	1497.88
	1050 - 1100	37.14	30.44	32.42	316.17
	1100 - 1200	45.06	28.02	26.92	606.43
	1200 - 1300	56.15	11.90	31.95	340.09
	> 1300	52.14	10.66	37.20	258.89
Distur-bance event	None	47.69	15.11	37.20	2911.14
	Fire	18.21	73.33	8.46	145.68
	Windfall	51.36	45.77	2.86	146.99

(Table 3-9). Increases in crown closure occurred most frequently in the absence of apparent disturbance events, while areas of unchanged crown cover were common in areas experiencing either windfall or no apparent disturbance at all (Table 3-9).

3.4 Discussion and Conclusions

3.4.1 General Trends of Change

Results indicate that there has been a general shift in the study area from early to later successional vegetation types. Young tree growth, forb, grasslands, and exposed rock decreased in prevalence, in favour of forest, shrub, and anthropogenic cover. Spatial analysis showed that several cover types which remained constant in absolute aerial extent on the landscape also underwent successional transitions. Two-thirds of open forest, for example, shifted to forest cover, but was replaced in part by grassland succeeding to open forest. Similarly, shrub cover which changed to forest was replaced by forb transforming to shrub. Sand/gravel cover was especially unstable; over three-quarters of it shifted to water, forb, wetland, or forest, while new sand/gravel occurred in areas formerly covered in water. Finally, although wetland areas appeared unchanged, almost half of the area shifted to forest, shrub, and water, while new wetlands occurred in areas of former water, shrub, forest, and sand/gravel. In addition to an absolute increase in forest area from 1949 to 1991, there was a significant increase in crown closure of existing forest. Finally, there appeared to be a general trend towards increased homogeneity on the landscape in 1991. In most cases, patch number declined and patch size increased by cover type or forest attribute.

Some of the apparent changes can be attributed to the airphoto interpretation process. The apparent decline in the amount of rock, for example, occurred as a result of the classification scheme rather than an acceleration of geological processes. The category "rock" was limited to those areas of rock substrate with less than 5% vegetation; as soon as the amount of vegetation increased beyond that threshold, the polygon was classified as something else, although the substrate was likely still rock. So while it is not strictly true to suggest that there was a decline in rock, it is correct to suggest that there has been an increase in vegetation cover on rocky areas. A similar case might be argued for transitions between water and sand/gravel bars. Distinctions between these two cover types are largely dependent on water levels, which are influenced by both time of year and recent precipitation trends. Both sets of airphotos were flown in mid-late September (Sept. 15, 1949 and Sept. 25, 1991), thus minimizing variation due to seasonal

fluctuations in water levels. Without detailed climate data, the effects of longer-range precipitation trends are hard to determine.

These caveats notwithstanding, the results are consistent with the hypothesis that the vegetation within the study site is aging, and has been relatively unaffected by those processes that reset the successional clock. The fire history of the area has been detailed elsewhere in this thesis and will not be repeated here, but research findings suggest that fire events in the park have been much less frequent this century than the preceding several hundred years (see Chapter 1 for further discussion), which may account for many of the observed changes. In the absence of fire, a process which typically acts to restart the successional sequence, we expect to find grass and forb cover transforming over time to shrub and forest, and forest crown closure increasing.

3.4.2 Forest Encroachment

Forest encroachment on grasslands has been reported in numerous areas, including the Colorado Front Ranges (Mast *et al.* 1997), Yellowstone National Park (Houston 1982), southwestern Montana (Arno and Gruell 1986), and northern Indiana (Cole and Taylor 1995). Several potential explanations have been suggested. In areas where climate potentially favoured forest establishment, frequent fire events in the past may have maintained grasslands and limited forest encroachment by killing tree seedlings (Mast *et al.* 1997). Decrease in fire frequency this century may also have favoured the establishment of trees (Mast *et al.* 1997, Arno and Gruell 1986, Houston 1982). Increases in grazing pressure on grasslands and rangelands with the introduction of livestock may have eliminated much of the fine fuels needed to carry fires, thus contributing to a decrease in fire frequency (Arno and Gruell 1986). On the other hand, it has been suggested that decreased browsing pressure on tree seedlings has facilitated forest encroachment elsewhere (Mast *et al.* 1997). Finally, where tree establishment is limited by moisture availability, changes in climatic conditions may favour increased forest encroachment (Mast *et al.* 1997).

The differential rates of forest invasion by aspect and proximity to adjacent forest observed in my study area are not straightforward to explain. Greatest rates of forest encroachment occurred on south slopes and within 100m of existing forest. The latter

observation is not unexpected, given that seed availability declines with the distance from its source. The former observation, however, is the opposite of what has been found in other areas. In both the Colorado Front Ranges (Mast *et al.* 1997) and northern Indiana (Cole and Taylor 1995), greater forest invasion occurred on moist north slopes. Forest encroachment was less pronounced on south slopes, probably limited by moisture availability (Mast 1997, Cole and Taylor 1995). It may be that moisture availability is not a limiting factor for forest encroachment in the study area. It is also possible that fire events in the past occurred more frequently on drier south slopes, thus restricting the encroachment of forest more on these slopes than elsewhere. The effects of the reduced fire frequency in this century may thus be more pronounced on south aspects, particularly if tree establishment is fastest in these warm areas.

3.4.3 Changes in forest structure and composition

Increases in average forest canopy closure presumably due to decreased fire frequency have been observed in Banff National Park (Achuff *et al.* 1996), southern interior British Columbia (Taylor and Hawkes 1997), the Colorado Front Ranges (Veblen and Lorenz 1991), and eastern Oregon and Washington (Lehmkuhl *et al.* 1994). Increases in canopy cover observed in this study area are likely due to a number of factors. Crown closure typically increases with stand age until the canopy starts to breakup as overstorey trees mature. In the virtual absence of fire events on the landscape over the last century, average stand age has been continually increasing. Field observation suggests that in most cases, stand break-up has not yet begun to occur, especially in the case of lodgepole pine stands, many of which date to the 1889 fire. There is also a high occurrence of stands with two layers of lodgepole pine in the montane ecoregion, with the older trees having survived the fire which gave rise to the younger cohort (Tande 1977, LaRoi and Hnatiuk 1980). It is possible that the maturation of the understorey cohort may have contributed to increased canopy closure. While it is not possible to infer anything about changing stand density from this study, other research suggests that tree density has declined in lodgepole pine stands in the montane over the last 22 years (Stadt 1993), a trend which has obviously not been accompanied by a decrease in canopy cover.

Decreases in forest canopy cover were correlated with areas in which disturbance by either fire or wind was apparent. It is possible that other low intensity disturbances not visible on the airphotos, such as ground fires, wind or insect damage, have also contributed to the decrease in crown closure in some areas. Such factors may also help to account for the 28% of open forest stands which did not increase in crown closure between the two dates. Decreases in canopy cover may also be due to stand break-up in some areas. In some cases, particularly in mixed coniferous forests containing Douglas-fir, stands classified as having a single-canopy structure in 1949 changed to a multiple canopy structure in 1991. Many of these stands are currently characterised by the occasional veteran Douglas-fir tree which penetrates through a fairly closed understorey layer of lodgepole pine and white spruce. The analysis of canopy cover in this study suggests that crown closure in these stands has decreased over time; while this may be true, this simple analysis masks the more complex transitions which are evidently occurring. Detailed analysis of these changes, however, is beyond the scope of this study.

Increases in crown closure were more likely to occur at lower elevations, while decreases in closure were more prevalent at middle elevations and on south and west slopes. It is possible that succession occurs most rapidly at lower, warmer, elevations, thus explaining the preponderance of increasing crown closure in these areas. It is also possible that fire frequency was previously highest at the lowest elevations in the montane. Fire exclusion over the last century may therefore have had the greatest effects at lower elevations. The prevalence of decreases in canopy cover at middle elevations may be explained in two ways. Historically, fire frequency decreased with elevation (Tande 1977). Thus older stands may occur at higher elevations, and stand break-up may explain the higher incidence of decreased crown closure. A more likely explanation involves the prescribed burn on the Colin Range, a large burn at middle elevations on south and westerly slopes, which may account for a large portion of the observed trend. To my knowledge, no other studies have attempted to examine forest crown closure changes by aspect or elevation, thus making comparisons with other areas difficult.

Successional trends in the species composition of forest stands are also interesting. Half of the area occupied by deciduous stands in 1949 was still dominated by deciduous species in 1991; the other half was dominated by coniferous species. It is

possible that the direction of change is correlated with stand age, with deciduous trees in older stands more likely to be replaced by conifers. More difficult to explain is the loss of the conifer component in 51% of deciduous/coniferous stands. Typically, the conifer component increases as deciduous stands age. It is possible that some of these areas experienced disturbance events which would have stimulated the suckering of aspen trees.

3.4.4 Increase in Homogeneity

The trend toward greater apparent homogeneity of cover types at the landscape scale is also an expected side effect of a low level of disturbance on the landscape. Fire, wind, and disturbance by insects typically produce complex patterns; patch size and shape on the landscape is a direct reflection of this disturbance history. In the absence of disturbance events, vegetation is expected to converge towards older successional stages with time, resulting in a more homogeneous landscape. This process has been described in other areas with a history of decreased fire frequency including eastern Oregon and Washington (Lehmkuhl *et al.* 1994), and gallery forests in Kansas (Knight *et al.* 1994). It is important to note, however, that although homogeneity may have increased at the landscape scale, this is not necessarily the case at the stand level. An exploration of heterogeneity at the stand level was beyond the scope of this project. Uneven age structures and complex species composition are characteristic of the montane ecoregion (Tande 1977), and would be worthy of more detailed study.

3.4.5 Conclusions

Overall, the results of the work supports the hypotheses outlined at the beginning of the work. Descriptive and spatial analysis of the study site indicates an increase in forest cover, and general trend toward later successional seral stages from 1949 to 1991. Forest vegetation has encroached onto grasslands, and there has been a general increase in crown closure in coniferous stands. Finally, there has been an increase in the homogeneity of cover types at the landscape level.

A comparison between these results, and those reported in Chapter 2 will be presented in the final chapter, as will an examination of the potential implications of these changes, and possible management options.

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CHAPTER 4

Bringing the Past to Bear on the Future

The vegetation in the montane ecoregion of Jasper Natural Park has been shaped for millennia by both the internal processes of successional change, and the external forces of nature - fire, wind, water – and culture – burning, cropping, logging, and so on. That this landscape is inherently dynamic is well-known. The details of these changing states, however, are less well understood. Although we have an idea of how fire regimes and other processes in the park have changed over the last few centuries, there is little detailed research on how vegetation patterns have changed in response.

The results of this study begin to fill this gap. The composite view provided by both the Bridgland historical photographs, repeat images, and airphotos depict a landscape that has shifted towards later successional vegetation types. Grassland, shrub, young regenerating forest stands, and open forests common on the landscape at the beginning of the century have declined in favour of closed canopy coniferous forests.

In this final chapter, I will begin by drawing together the results of the Bridgland repeat photography and the airphoto interpretation to provide a composite picture of vegetation change in the montane ecoregion over the last eighty years. The potential reasons for this change will be considered, as will the ecological implications for the ecoregion as a whole. Finally, I will introduce the concept of reference ecosystems and suggest ways in which this work may help us to decide on a future course of management for Jasper National Park.

4.0 Eighty Years of Change

The analysis of both the Bridgland repeat photographs (Chapter 2) and the airphotos (Chapter 3) document substantial change in the montane ecoregion of Jasper National Park over the last 80 years. Both showed a dramatic increase in the overall

extent of forest cover and degree of forest canopy closure, and a decrease in the number of deciduous and young regenerating stands. Forest encroachment into grasslands was apparent in both sets of data, as was a trend toward less patchy, more homogeneous vegetation cover over the area. Finally, there was a significant increase in anthropogenic cover in the study area.

4.0.1 Evaluating the Oblique Photo Interpretation Methodology

In order to determine if the results of the two approaches could be compared, it was necessary to evaluate the methodology used to quantify change in the repeat photographs. Recall that the Bridgland photographs were oblique. The scale of the photographs thus varied within a given photograph, making calculations of absolute area very difficult. This difficulty notwithstanding, I developed a simple method for analyzing the photographs quantitatively which I hoped would provide some approximation of the degree of change within the study site between 1915 and 1997. The airphotos, of course, posed no such geometrical problem – because they were vertical photographs, absolute measures could be determined fairly easily once they were orthocorrected. By comparing the relative values of the various vegetation types in 1997, as determined on the oblique photographs, with the absolute values as determined from the 1991 airphotos, it was possible to estimate the accuracy of the oblique interpretation methodology.

The two methodologies resulted in strikingly similar values (Table 4-1). The area estimations in 1991 and 1997 for 10 of the 13 cover types [Forest(B), Forest(C), Forest(D), young tree, shrub, grassland/herb, wetland, water, sand/gravel, and rock] were within 2% of one another (Table 4-1). The largest discrepancy was for Forest(A) cover, which was calculated to be 10% in the 1991 airphotos, and 4% in the 1997 oblique photographs. An opposing trend was observed for Open Forest, however, the airphoto and oblique photograph estimates were 2% and 5% respectively. Because Forest(A) and Open Forest differ only by one crown closure class, there may have been an inconsistency in differentiating between the two categories. Quantifying crown closure from oblique views – where forest stands are frequently viewed from the side – is probably more difficult than doing the same from vertical airphotos, where stands are

Table 4-1: Vegetation cover in 1915, 1949, 1991 and 1997. The 1915 and 1997 data are relative photograph areas (%) as calculated from the Bridgland oblique photographs (c.f. Chapter 2). The 1949 and 1991 data are absolute areas (%) calculated through standard airphoto interpretation techniques (c.f. Chapter 3). Comparing the 1991 and 1997 data permits an evaluation of the accuracy of the oblique interpretation methodology.

	1915 Bridgland	1949 Airphotos	1991 Airphotos	1997 Bridgland
Forest (A)	15	4	10	4
Forest (B)	16	13	9	10
Forest (C)	3	27	24	23
Forest (D)	0	13	21	23
Open Forest	16	2	2	5
Young tree	7	4	0	0
Shrub	13	4	6	4
Grassland/Herb	11	9	4	6
Wetland	2	4	3	2
Water	12	13	12	13
Sand/Gravel	2	2	1	1
Rock	2	4	3	2
Anthropogenic	1	2	4	8

viewed from overhead. Combining the data from the two cover types together would lessen the discrepancy between the oblique and airphoto estimates.

The other large difference observed was for the Anthropogenic category (4% in the airphotos, 8% in the obliques) (Table 4-1). This difference was likely due to the genuine geometric problem with trying to quantify oblique photographs. One set of the medium-scale Bridgland photographs was taken from a ridge that today overlooks an area of concentrated human activity, including the power-generating station, the dump, and an old gravel extraction pit (c.f. Plate 2-2, Plate 2-3). These human structures dominate the foreground of these pictures, and their relative percent area is thus biased upwards. I had hypothesized that with a large random sample of photographs, these biases would compensate for one another. It appears that this is the case overall, except for the particular case of anthropogenic cover.

The significance of the congruence between the results of the two methodologies is quite important. If the oblique interpretation methodology provided a good estimate of the absolute area in the present, then the relative values calculated for the 1915 landscape

are most likely a good estimate of absolute values then. Thus the two studies are directly comparable, and together provide a good idea of the changes that have occurred on this landscape over the last century.

4.0.2 Combining the two data sets

As noted previously, the same general trends of vegetation change are evident in the two sets of data. Further comparison permits a more detailed interpretation of when the observed changes occurred. For forested areas, Forest(A) and Open Forest declined greatly from 1915 to 1949 (Table 4-1). The trajectory of change from 1949 to the present is a little harder to discern, but it appears that there may have been a slight increase in these stand types. The initial rapid decrease may be evidence of a landscape recovering from the effects of the widespread fires of 1889. Moreover, a second wave of fire which passed through some of the area in 1905 may have resulted in a second cohort of young seedlings and stands not yet detectable in the 1915 photographs (Tande 1977). Increased crown closure may thus be a combination of both increased stem density and increased individual crown size. The observed increase in Forest(A) and Open Forest stands post-1949 may be due to several factors: the introduction of prescribed burns and several large windthrow events which would have resulted in mortality in existing stands, and the encroachment of new open forest on formerly unforested areas. Area occupied by Forest(B) decreased in equal amounts in the first and second halves of the century. Area occupied by Forest(C) increased greatly in the early part of the century, and decreased slightly after that point (Table 4-1). Finally, Forest(D) increased in area throughout the eighty year period.

For non-forested areas, net shrub cover declined dramatically in the first part of the century and then remained stable. Grassland vegetation declined more after 1949. The reasons for this are not clear. There did not appear to be an increase in forest encroachment after that date (c.f. Table 2-1 and Table 3-5). However, no differentiation was made between grassland and forb categories in the oblique photograph interpretation, therefore separating the trends in forest encroachment on grasslands and natural succession patterns in forb communities is difficult. Young tree growth declined throughout the eighty year period. Absolute change in anthropogenic cover was hard to

determine because of the possible bias explained above, but increased up to eight times over the period examined. Net wetland, water, and sand/gravel cover appeared to remain constant over the period. It is hard to assess the reliability of this finding however, since these systems are inherently dynamic and water levels, in particular, fluctuate throughout the season.

The Bridgland analysis suggested that few stands overall experienced a decrease in crown closure from 1915 to 1997; this phenomenon was more common (in all canopy closure categories) from 1949 to 1991, based on the airphoto analysis. Methodological differences are probably responsible. First of all, a greater level of detail (resolution) in the airphoto classification likely permitted the detection of more subtle changes. Secondly, crown closure may have been overestimated in the oblique views since stands are viewed from the side rather than the top.

4.1 Drivers of Change

While it is recognized that ecosystems are inherently dynamic, park managers and ecologists are worried that the current state of the montane ecoregion of Jasper is outside the natural range of variability expected in the area (Achuff *et al.* 1996). Jasper's National Park status has afforded it general protection from the large-scale obvious impacts common on the landscapes which surround it. However, it has been affected by more subtle forms of change, both natural and anthropogenic.

4.1.1 Anthropogenic Activity

The results of both the oblique and aerial photograph interpretation suggested that there has been between a four-fold and eight-fold increase in anthropogenic cover in the study area over the last century. These numbers alone are disturbing, and a careful consideration of the nature of these changes suggests even greater cause for concern.

To begin with, the study area was not representative of the anthropogenic activity which has occurred in the montane ecoregion as a whole in Jasper. Human activity in the montane is highest in the Three Valley confluence area – the region where the Athabasca,

Maligne, and Miette rivers converge. The townsite of Jasper is located in this area, as are numerous outlying commercial accommodations, campsites, trails, and other tourist infrastructure. The study area was located outside of this epicentre of human activity, thus the change in anthropogenic activity over the last century reported here is not representative of the montane ecoregion as a whole, and was probably an underestimation of the magnitude of change.

Secondly, the effects of the increase in anthropogenic cover reach far beyond the boundaries of the individual anthropogenic sites. The power generating station, for example, also generates much noise pollution, which may affect wildlife behaviour in the area. Air-borne and water-borne pollutants are often associated with anthropogenic sites; effects of these are dispersed far from the source of the pollution.

Finally, while it was possible to classify the changes in anthropogenic cover in the study area, the methodologies employed did not allow for the documentation of the attendant changes in human activity on the landscape. For example, while the highway was present in both the 1949 and 1991 airphotos, the number of vehicles travelling along this corridor has increased over time, as has the annual road-kill. Similarly, the area today occupied by the Palisades Environmental Training Centre was a homestead occupied by the Swift family in 1915. While the physical footprint of these human occupations may be similar, the Palisades Centre today hosts a far larger number of people. Increased human activity on both the site and the surrounding trails may result in increased wildlife displacement not visible through airphoto interpretation.

The effects of this increase in human activity on the vegetation in the montane ecoregion are hard to quantify. In addition to the direct loss of habitat, anthropogenic activity such as the building of retaining walls along the Snaring River, and the fragmentation of the Esplanade wetland by transportation corridors, has impacted the vegetation in the area. Increased human activity can also have indirect effects, for example by facilitating processes such as the spread of non-native species (Achuff *et al.* 1996).

4.1.2 Fire

It is generally accepted that the extent, frequency, and intensity of fire events are responsible for shaping much of the vegetation patterns in Jasper. Fire events, in turn, are influenced by both physical landscape patterns, such as topography, by antecedent vegetative characteristics like species composition, stand age, and stand condition, and by climatic patterns (Rogeau 1996, Weber and Flannigan 1997). Fire history records for Jasper National Park, and in fact for many areas in the Rocky Mountain region, suggest that there has been a dramatic shift in the fire regime over the past hundred years (Tande 1977, Van Wagner 1995). Until 1913, the mean fire return interval in the montane ecoregion of Jasper was between 17 and 26 years (Tande 1977). The last fire of any magnitude (other than prescribed burns) in the park occurred in 1908. No fire free period of this magnitude – 90 years – has been reported previously in the fire record of the park (Tande 1977).

The reasons for the change in fire frequency are much debated. The efficiency of modern fire suppression techniques is usually cited as the prime reason. However, fire suppression techniques did not gain any real degree of efficacy until the 1940's, and although today we are relatively good at putting out small fires, the effectiveness of fire suppression techniques diminishes rapidly as fires grow in size – and it is the large fires that are responsible for the majority of area burned (White 1985, Masters 1990). It has also been suggested that the influx of European settlers in the late 19th century was accompanied by an increase in human ignited fires, thus biasing the apparent fire frequency upward (Byrne 1964, Nelson and Byrne 1966, Tande 1977). Detailed examination of the fire record in several mountain jurisdictions, however, did not find a pattern of increased fire frequency during the period of European settlement (Johnson and Fryer 1987, Johnson and Larsen 1991, Johnson *et al.* 1990, Masters 1990, Van Wagner 1995, White 1985).

A recent examination of lightning patterns has prompted the suggestion that there may not be sufficient lightning activity in the Jasper region to account for the historical fire frequency (Heathcott 1996). The park appears to lie in a 'lightning shadow': storm systems arriving from the west lighten their loads as they mount the continental divide, and erupt with renewed vigour only once they reach the foothills east of Jasper (Heathcott

1996). The missing ignition source might be the management practices of the First Nations peoples who once lived and travelled in the area. The number of First Nations peoples, the extent of their management practices, the ways and frequencies with which they used fire, and the degree to which they shaped this landscape, are all matters of great contention (Lewis 1980, Barrett 1981, Kay and White 1995, Johnson and Larsen 1991). Specific research from the boreal forest of Alberta and more general research from the United States suggests that aboriginal peoples used fire extensively to manage the landscape (Lewis 1980, Pyne 1997). Métis use of fire in the late 1800s has been documented in Jasper (Murphy 1980), but the general extent and ecological importance of First Nations use of fire in the Rocky Mountains is an issue of great contention (Kay and White 1995, Barrett 1981, Heinselman 1975, Johnson and Larsen 1991).

The contribution of changing climate patterns to the fire regime in Jasper is also uncertain. Many of the key characteristics of fire regime, including frequency, size, and intensity, are highly dependent on climatic conditions (Weber and Flannigan 1997). Warmer temperatures and lower precipitation may decrease fuel moisture content, extend fire season, and increase fire frequency and severity (Weber and Flannigan 1997, Johnson *et al.* 1990). Fire history studies conducted in the southern Canadian Rockies (Johnson and Larsen 1991), Kootenay National Park (Masters 1990), and Glacier National Park (Johnson *et al.* 1990), suggest fire frequency decreased during the Little Ice Age, a period of cooler, moister weather which began shortly after 900 BP and culminated in the late 18th and early 19th centuries (Osborn and Luckman 1988, Luckman 1990). Research in the southern boreal forests of eastern Canada, however, found the opposite pattern: an increase in the fire cycle with the onset of warmer (but moister) climate following the Little Ice Age (Bergeron and Archambault 1993). Interpretation of the fire record in Jasper suggests a single fire cycle between 1533 and 1915 of about 110 years; there was no observed change in response to climatic change during the Little Ice Age (Van Wagner 1995). It would appear therefore, that the relationship between fire regime and climate is either highly regional or questionable. The climate record suggests increased temperatures in Jasper this century following the end of the Little Ice Age (Osborn and Luckman 1988), but the potential contribution of warmer weather to the observed changes in fire regime are difficult to ascertain.

That there has been a major change in the fire regime in the montane ecoregion of Jasper this century versus the preceding several centuries is clear. To what degree these changes are the result of the dynamics of naturally occurring processes or to the mindful, or not-so-mindful, interventions of human beings, however, is not clear. Disentangling the confounding effects of changing climate, the arrival of European settlers, the displacement of Native peoples and their management practices (and how significant these might have been), and the implementation of the policy of fire suppression, is extremely difficult, if not impossible. While we may never fully sort out the reasons for the changes in the fire regime, the implications of continued fire exclusion on the vegetation in the montane are becoming increasingly obvious.

4.2 The Implications of Continued Fire Exclusion

One of the greatest changes in the montane ecoregion over the last century is the overall decrease in vegetational diversity on the landscape. In Banff National Park, where a similar story has unfolded, parks managers predict that if the current fire regime persists into the future, fully one-third of vegetation types may be lost entirely from the landscape (Achuff *et al.* 1996). Aspen stands, in particular, are at high risk, as are open Douglas-fir stands and grasslands (Achuff *et al.* 1996).

The loss of these systems translates into decreased habitat availability for several important wildlife species. Grizzly bears, for example, rely heavily on buffaloberry (*Sheperdia canadensis*) crops for much of their diet in the late summer (Achuff *et al.* 1996). Since berry production is negatively correlated with crown closure, grizzly bears have been negatively impacted by increasing forest canopy cover (Hamer 1995). Similarly, as the spruce component in ageing lodgepole pine stands increases, palatable forage species such as understorey shrubs and grasses are gradually replaced by mosses (Achuff *et al.* 1996). Finally, aspen stands house more abundant and diverse assemblages of birds and understorey plant species than any other forest type (Achuff *et al.* 1996). The loss of these stands will be accompanied by a decline in overall park biodiversity.

The decrease in fire frequency over the last ninety years may also be sufficient to permanently alter the fire regime in the park (Heinselman 1975). The montane ecoregion

had a history of frequent low-intensity fires until early this century. Frequent fire would have limited the build up of fuels, thus decreasing the possibility of large scale intense fire in most places. Given 90 years of fuel accumulation today, it is possible that should a fire be initiated in the Athabasca Valley, the resulting blaze would be far different from what was experienced in the past, and could in fact change the fire regime to one characterized by less frequent higher intensity burns (Heinselman 1975, Barrett 1996). The ecological and social consequences could be quite dramatic, on par with those seen in Yellowstone a decade ago.

Furthermore, the increase in homogeneity of forest stands may also facilitate the spread of insect disturbance. Outbreaks of spruce-budworm in the boreal forest, for example, are most severe in continuous mature and over-mature fir-spruce stands (Van Ralte 1972, Blais 1983). Where stands are interspersed with patches of other types of vegetation, insect damage is much less extensive. The combination of increased continuity and age of forest stands in the montane ecoregion may thus result in unusually high susceptibility to insects such as the mountain pine beetle.

The potential ramifications of continued fire exclusion – decreased biodiversity, the loss of wildlife habitat, and the potential for both widespread fire and increased insect outbreak - in the montane ecoregion of Jasper are serious, especially in a national park mandated to protect ecological integrity. Parks managers, acknowledging the gravity of the plight, are beginning to search for solutions. Increasingly, it is to the past that they are looking. The field of ecological restoration may offer some answers.

4.3 Does the past reflect the future?

The results of my study provide, for the first time, detailed documentation of past vegetation conditions in the montane ecoregion of JNP. Why is this important? In a landscape that has experienced significant perturbation during the 20th century, knowledge of past conditions can help to guide future management practices.

The concept of reference ecosystems has been proposed as a useful framework to guide the work of ecological restoration (Aronsen *et al.* 1995). While the need for

restorative practices is often obvious in a given ecosystem, the goals of that restoration are usually harder to define. Quantifying reference conditions – the range of historic variability in the ecological structures and processes of a system – can provide both a measure of the current state of an ecosystem, as well as goals for restoration treatments (Fule *et al.* 1997).

Reference data can be gathered from several sources (White and Walker 1997). It is sometimes possible to find analogous ecosystems elsewhere on the landscape which have escaped perturbation. Where this is not possible, historical data predating the perturbation can provide insight. Jasper National Park was set aside in part to shelter the landscape from the effects of rampant human activity. As such, it is often used as a reference to measure ecosystem function in areas subject to greater human pressures. If the park is now in need of restoration itself, to which reference can we turn?

The results of this work are the first step in quantifying the historical range of variability of vegetation patterns in the montane ecoregion of the park. While there has been general consensus that the current fire regime is outside of the range of variation observed over the last four centuries (Achuff *et al.* 1996), knowledge of the accompanying changes in vegetation patterns is lacking. We simply don't know if the current state of vegetation in the park is outside the historical range of variability. Analysis of soils on certain grassland sites in the montane, for example, shows that Eutric Brunisols characteristic of forested areas were once dominant (Holland and Coen 1982). Thus what is today grassland was likely once forest, and the observed forest encroachment that is currently occurring may be well within the historical range of variability.

Much work remains to be done. While it is evident that the current fire regime is unprecedented over the past four hundred years, knowledge of conditions previous to that time is still lacking. Moreover, more detailed information on the changes which have occurred over the last few centuries is also needed. Stand reconstruction techniques, for example, can be used to document past forest density and species composition (Fule *et al.* 1997). The work must also be extended beyond the boundaries of the montane ecoregion. Although research effort has been focused in the valleys of Jasper, subalpine and alpine regions have not been exempt from change.

Applied research into potential restorative treatments is also needed. The introduction of prescribed burns has now been mandated in national parks in Canada. Emulating historical fire regimes, however, is more complex than simply dropping a match. Reintroducing fire onto the landscape requires a consideration of the effects of time of year and intensity of burn, in addition to extent and periodicity of fire events (Baker 1994, Johnson and Miyanishi 1995). Moreover, the reintroduction of fire after a long period of exclusion does not always have the desired effects. Stand thinning and other treatments may be necessary in addition to prescribed burns to restore former conditions (Feeney *et al.* 1998). Douglas-fir stands, in particular, may require special consideration in a restoration context. Accumulated fuels may need to be removed from around veteran trees before fire is re-introduced. The issues of how best to re-establish and maintain both open-grown Douglas-fir stands, and Douglas-fir as a component of closed-canopy coniferous stands, also need attention.

Careful long-term monitoring of any restorative treatments is also essential. The establishment of permanent plots and the consistent collection of data from these sites will go a long way to helping monitor the effects of our experiments.

Finally, research into historical human activity and the impacts of increased development on the landscape is also necessary, as is careful reflection of how this knowledge can be incorporated into decision making. While the reasons for the decrease in fire frequency this century are unclear, it seems likely that the displacement of First Nations peoples from the park was a factor. If it turns out to be true that much of the historical fire regime is a result of extensive fire management on the part of aboriginals, will we still view fire in the same way?

At a certain point, decisions about ecological integrity pass from the scientific into the evaluative. And it is perhaps this challenge which is the greatest of all. Science can inform decisions about how to manage these ecosystems – we can use it to infer wildlife responses to decreased habitat availability, to suggest effects of fuel loads on future fire intensity, to recommend management options to increase the health of aspen stands, but it cannot make decisions. At a certain point, we must make a decision about what we value in Jasper National Park and use these values to define management objectives. This process will require not only detailed historical, ecological, and cultural

research, but a genuine commitment to raising, articulating, and arbitrating the values of all interested players (Higgs *et al.* 1998).

4.4 Mapping the future

The results of this study document eighty years of vegetation change in the montane ecoregion of Jasper National Park. In the virtual absence of fire this century, there has been a shift towards late successional vegetation types on the landscape and an increase in crown closure in forest stands. Grasslands, shrub, young tree growth, and open forests have decreased in extent, and closed canopy coniferous forests have become more prevalent. Anthropogenic cover has increased four to eight-fold.

The implications for ecosystem structure and function are potentially quite serious, and include decreased vegetation diversity, decreased habitat quality for several important wildlife species, and the potential for both widespread high-intensity fire and insect outbreak. The results of this work can be used both to define historical reference conditions and to help establish restoration goals for the montane ecoregion of the park.

When M.P. Bridgland arrived in Jasper National Park in 1915, he was charged with the creation of the first topographic map of the park, to show people the wonders that lay amidst the majestic peaks of the great Rockies. Today it is time to reach back into the past and return to that work of mapping, mapping not the present, but the future.

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APPENDIX 1

Historical and oblique photographs used for analysis in Chapter 2.

Survey station and photograph numbers refer to those used to classify the original images taken by M.P. Bridgland in 1915 as part of the Photo-topographical survey of the Central Part of Jasper Park. A copy of the original images is held in the Jasper National Park Warden Library. A copy of the repeat images will be deposited in the University of Alberta Archives.

Qualitative analysis only

Survey Station	Photo #
No. 34 – Roche Bonhomme	280
No. 39 – Morro Peak II	314
No. 40 – Hawk Mountain	318
	321
	323
	327
	328
No. 56 – Pyramid Mountain	448
	452
	453
	454
	457
No. 57 – Power House Cliff	462
	463
	464
	465
No. 58 – Henry House Flats	466
	467
	468
	469
No. 59 – The Palisade	472
	478
No. 60 – Snaring River	482
	483
	484
	485
No. 62 – Esplanade Mountain	503
	505
	507
No. 63 – Greenock Mountain	508
	511
	513
	514

Qualitative and quantitative analyses

Survey Station	Photo #
No. 34 – Roche Bonhomme	274
	275
	282
No. 38 – Morro Peak I	308
	309
	310
	311
	312
No. 39 – Morro Peak II	313
No. 40 – Hawk Mountain	317
	319
	320
No. 57 – Power House Cliff	459
	460
	461
No. 59 – The Palisade	470
	471
	481
No. 62 – Esplanade Mountain	504
	506

APPENDIX 2

Qualitative Interpretation of Oblique Photographs – A Preliminary Methodology

The Bridgland Topographical Survey

The historical photographs used in this study were taken as part of a phototopographical survey conducted from June – October, 1915, by M.P. Bridgland, a Dominion Land Surveyor working for the Canadian Department of the Interior (Bridgland 1924). The photographs are unique in the sense that they were taken specifically to document the Jasper landscape. Thus in addition to providing systematic and comprehensive coverage of the area, they are organized in a way that facilitates rephotography.

Bridgland set up a total of 93 survey stations in the north-central part of what is today Jasper National Park. The majority of the stations were established on the tops of mountains or prominent ridges and provide panoramic views of the surrounding area. A number of stations were also set up at ground level along the railroad tracks to survey the railway line. A total of 750 pictures were taken. Most stations consist of 8-12 photographs which circle the entire horizon. Photography was most likely done with a large format camera, 4¾" x 6½" glass plate negatives, 164 mm Zeiss Tessar Series III lens, B/W panchromatic emulsion, and a Wratten and Wainwright "G" filter (yellow) (Bridgland 1924). The photographs were used to create a series of topographical maps of the area. These are available as both six sheets (scale of 1 inch: 1mile) which can be edge matched to cover the north-central part of the park, and as one smaller summary sheet which covers the entire area. The field notebooks and original glass plate negatives have either been misplaced or lost entirely; despite intensive searching they have not yet been located.

A complete set of the original photographs is stored in the Jasper National Park Warden Library. These are organized by survey station and bound in a series of books. A copy is also deposited with the Jasper Yellowhead Archives and in the National Archives of Canada. Copies of the topographical maps are available in both the Jasper National Park Warden Library and in the rare map collection at the University of Alberta.

A number of other photo-topographical surveys were conducted by M.P. Bridgland and others, including the celebrated A.O. Wheeler, in the mountain regions of Alberta and British Columbia at the turn of the century, and may prove useful in reconstructing past landscape states.

Repeating the Photographs

Much has been written about the technique of repeat photography (Rogers *et al.* 1984). In essence, it consists of relocating the point from which the original image was taken, and

rephotographing the subject. Ideally the original camera equipment and film, time of year, and even time of day are also replicated as closely as possible.

Using the original topographical maps as a guide, I selected the stations which seemed most likely to have images which would depict the study area. I then browsed through the photographs taken at each of these stations, noting all of the ones that showed parts of the study area. High resolution laser black and white photocopies of these photographs were made for the field.

Relocating the general location of the survey stations was relatively straightforward using the original topographic maps. Serious scrambling was required to reach many of the stations, but few necessitated technical climbing skills. In many cases, Bridgland erected a large stone cairn at the main survey location which he could then site and take bearings from at future survey stations. These are a welcome sight after a long climb! Although Bridgland appeared to take the majority of the photographs from the main survey location, it was not unusual for secondary survey locations to be set up in the vicinity – usually to provide a better view of a particular section of the panorama. Determining the exact number and position of photograph locations at each survey site was therefore not always easy.

Once the general area was located, the photographs were divided into two groups – those which depicted foreground and those which did not. Foreground objects were vital in determining exact survey locations. In many cases, large rocks evident in the 1915 photographs were still present and apparently in the same positions. By moving around until the relocated foreground objects lined up with other visible features in the photographs, the exact photograph locations could usually be determined. This was easiest to do where there were a number of photographs with foreground objects which could be used simultaneously to locate the right spot. It is important to note that movements on the scale of centimeters in all three dimensions could make a big difference in lining up the pictures. While such locational error results in very minor geometrical differences between paired photographs, the visual differences, especially in the foreground, can be quite acute.

I used a Linhof Technika 4x5""large-format camera, 90 mm Schneider-Kreuznach Angulon 1:6.8 lens, and Manfrotto tripod. A No. 85C Wratten filter (pale orange) was used to cut haze and increase contrast, although in a subsequent field season I substituted a polarizing and haze filter with much the same results. All photographs were shot with T-max 100 (black and white) film. Although the use of this equipment entailed both heavy packs and more finicky photography procedures, the resulting 4x5" negatives were well worth the effort.

Careful field notes were taken to document both the exact locations of the photographs, the time of day, weather conditions, and general notes that might be useful in subsequent rephotography. In some cases, photographs of the camera location were taken (with a 35 mm camera) to document exact locations.

For safe travel and portering, I relied on at least one field assistant at all times. Assistance varied from the highly-skilled to those who volunteered to haul gear. On longer, multi-day trips, a three or four person team would be ideal to ensure base camp provisioning and safe conditions for all stages of the photography.

Darkroom procedures

The negatives were developed commercially. The prints were all done by hand. In the darkroom, I placed the field photocopy of the original image on the printing easel, and focused the repeat photograph on top of it. I manipulated the height of the enlarger until prominent lines in the projected image – the horizon, mountain peaks, railroad lines – matched those on the photocopied original. Printing the image on paper cut to the same size then resulted in an image that was cropped to both the same size and scale of the original. Ilford RC Multigrade Paper and Ilford chemicals were used for printing, and standard darkroom procedures were followed.

Interpretation and Analysis of the Photographs

Photographs taken at ground level or at very high elevations were eliminated from the quantitative interpretation. At ground level, features in the foreground dominate the image, and the elevation is not sufficient to provide a good overview of the area. At very high elevations, interpreting ground cover becomes increasingly unreliable, and detectable patch size becomes so large as to be of questionable utility.

Photographs were covered with transparent acetate overlays. For each pair of images, 8-12 point features which could be accurately identified on each picture, and which were well-distributed throughout the picture area, were identified. Features included pointed mountain tops, rocky outcroppings, bridges and other long-lived human features, and occasionally less reliable features such as tips of islands which appeared to be unchanged. Features were marked as a point and numbered on each of the photographs in the pair – these were the ground control points.

The repeat acetate was then laid over a standard piece of graph paper, and the x,y coordinates (in millimeters) of each ground control feature were determined. The assumption was that if the ground control features were well chosen, and the repeat print was indeed geometrically similar to the original, then the coordinates could be used to overlay the two images. Thus an arbitrary coordinate system was established for each pair of photographs. Coordinates were measured from the repeat acetates and not the original acetates, so that the control features could be more easily identified in the field should this ever be necessary.

Interpretation of cover types was completed in a manner similar to standard airphoto interpretation. Areas of homogenous cover (compositionally and structurally) were delineated with the aid of a 8x magnifying loupe. Polygons were labeled as per the established classification system.

Once the photographs were interpreted, the acetates were digitized in the standard manner. An arbitrary coordinate system was used (0,0 in the bottom left hand corner, to approximately 18,18 cm in the upper right hand corner, depending on the size of the photographs). Coordinates of the control features were used to situate each photograph in the pair in this coordinate system. Overlaying the two images was then possible. Accuracy of overlaying the two images in each pair was assessed quantitatively using the residual mean average (RMA) of the control points in the digitizing package. Thus RMA for the repeat acetate was expected to be very low, since the coordinates were measured from it. If the RMA of the original acetate was significantly higher, then the reliability of the selected control features was assessed. If features themselves seemed reliable, but there was still high error in the location of the points, then it was assumed that there were geometrical differences between the photographs. Qualitative assessment of the accuracy of the overlays was possible simply by seeing how well obvious landscape features lined up. In one set of photographs, for example, a current road feature was apparently located in the middle of a river in 1915! A new image in this case was reprinted in the darkroom, which solved the problem.

Vector coverages were rasterized and imported into a Geographic Information System. A spatial cross-tabulation (transition matrix) was calculated for each pair of images. Results of these individual cross-tabulations were added in pixel units to create a summary table. Percentage photograph area occupied by each cover type was calculated from these pixel counts (c.f. Figure 2-2). A summary transition matrix was also created from these summary pixel data. For each 1915 vegetation type, percentage of that vegetation type changing to other types was calculated (c.f. Table 2-1).

Significance of the Results

It is important to recognize that the results of this approach do not represent absolute values on the landscape. They are a measure of the relative photograph area by cover type, and should not be advertised as anything else. That said, results of this study suggest that relative photograph areas calculated from the Bridgland photographs provide a good estimation of the actual ground area occupied by cover type in the study area. Further testing of the methodology is required, but preliminary results are hopeful.

Potential avenues for improvement

Work on the Bridgland repeat photography project continues, and several modifications are anticipated for future work. In order to facilitate future rephotography at these same sites, permanent markers (potentially small stakes with metal tags?) should be placed at all camera locations. Locating exact camera locations is by far the most time-consuming part of the work, and without the original fieldbooks, it is hard to be sure that we have found the exact spots. Better documentation, including photographs of the camera locations, and permanent markers would facilitate this process in the future.

In may also be worth considering the use of colour film. I used black and white film both for financial reasons and so that the images would be easily comparable. Barring financial constraints, it would be worth taking each repeat image in both colour and black and white.

Finally, the interpretation process becomes easier as the size of the images increases. I was limited to the existing set of archival images, since the original negatives were not available, which are approximately 4"x6". It would be worth, however, working with larger prints if possible.

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APPENDIX 3

Classification scheme for Airphoto Analysis

Each polygon was classified by cover type. Forested stands were further classified by stand structure, crown closure, and species composition. Modifiers were noted where applicable. Classification scheme was adapted from the Alberta Vegetation Inventory and Ecological (Biophysical) Land Classification of Banff and Jasper National Parks (see Chapter 3 for references).

Cover Type

Vegetated (>5% vegetation cover)
Forested (>6% tree cover):
- Forest (16-100% crown closure)
- Open Forest (6-15% crown closure)

Non-forested (<6% tree cover):
- Young tree growth (sapling stage)
- Shrubs
- Herb – Forb
- Herb – Grassland
- Herb – Wetland

Non-vegetated (<5% vegetation cover)
Water
Rock
Sand/Gravel
Anthropogenic

Species Composition

C – >80% coniferous species, <20% deciduous
CD – 60-80% coniferous, 20-40% deciduous
M – 40-60% coniferous, 40-60% deciduous
DC – 60-80% deciduous, 20-40% coniferous
D – >80% deciduous, <20% coniferous

Stand Structure

Single Canopy
Multiple Canopy
Complex Canopy

Crown Closure

Forest(A) – 16-30% cover
Forest(B) – 31-50%
Forest(C) – 51-70%
Forest(D) – 71-100%

Modifiers

Clearcut
Burn
Windfall

APPENDIX 4

Results of statistical tests for mean patch size by cover type, species composition, stand structure, and crown closure.

1. Cover Type

ANOVA[a,b]

			Unique Method				
			Sum of Squares	df	Mean Square	F	Sig.
LGAREA	Main Effects	(Combined)	35.313	11	3.210	7.330	.000
		TYPE	35.305	10	3.530	8.061	.000
		YEAR	.148	1	.148	.338	.561
	2-Way Interactions	TYPE * YEAR	5.344	9	.594	1.356	.204
	Model		40.624	20	2.031	4.638	.000
	Residual		377.530	862	.438		
	Total		418.154	882	.474		

a. LGAREA by TYPE, YEAR

b. All effects entered simultaneously

Independent Samples Test T-test for Equality of Means			
TYPE	**t**	**df**	**sig (2-tailed)**
Forest	-1.852	181	0.066
Open Forest	-0.321	75	0.749
Young growth	-	-	-
Shrub	-0.27	95	0.788
Grass	1.657	109	0.1
Forb	-0.377	71	0.707
Wetland	-1.05	70	0.297
Water	-0.522	98	0.603
Sand/Gravel	-0.853	53	0.397
Rock	1.625	74	0.108
Anthropogenic	-0.304	22	0.764

2. Species Composition

ANOVA[a,b]

			Unique Method				
			Sum of Squares	df	Mean Square	F	Sig.
LGAREA	Main Effects	(Combined)	9.714	5	1.943	3.923	.002
		YEAR	1.976E-03	1	1.976E-03	.004	.950
		COMP	8.972	4	2.243	4.530	.001
	2-Way Interactions	YEAR * COMP	1.934	4	.484	.977	.421
	Model		11.978	9	1.331	2.688	.005
	Residual		151.042	305	.495		
	Total		163.021	314	.519		

a. LGAREA by YEAR, COMP

b. All effects entered simultaneously

Independent Samples Test T-test for Equality of Means			
COMPOSITION	**t**	**df**	**sig (2-tailed)**
Coniferous	-1.826	184	0.069
Conif/Decid	1.235	22	0.23
Mixed	-0.771	30	0.447
Decid/Conif	0.14	28	0.89
Deciduous	0.877	41	0.386

3. Stand Structure

ANOVA[a,b,c]

			Unique Method				
			Sum of Squares	df	Mean Square	F	Sig.
LGAREA	Main Effects	(Combined)	2.635	3	.878	1.512	.212
		STRUC	1.154	2	.577	.993	.372
		YEAR	.889	1	.889	1.530	.217
	Model		2.635	3	.878	1.512	.212
	Residual		134.811	232	.581		
	Total		137.447	235	.585		

a. LGAREA by STRUC, YEAR

b. All effects entered simultaneously

c. Due to empty cells or a singular matrix, higher order interactions have been suppressed.

Independent Samples Test T-test for Equality of Means			
STRUCTURE	**t**	**df**	**sig (2-tailed)**
Single	-1.334	195	0.184
Multiple	0.124	35	0.902
Complex	-	-	-

4. Crown Closure

ANOVA[a,b]

			Unique Method				
			Sum of Squares	df	Mean Square	F	Sig.
LGAREA	Main Effects	(Combined)	18.131	5	3.626	9.352	.000
		YEAR	2.588	1	2.588	6.675	.010
		DENSITY	15.448	4	3.862	9.961	.000
	2-Way Interactions	YEAR * DENSITY	1.933	4	.483	1.247	.290
	Model		19.865	9	2.207	5.693	.000
	Residual		255.121	658	.388		
	Total		274.987	667	.412		

a. LGAREA by YEAR, DENSITY

b. All effects entered simultaneously

Independent Samples Test T-test for Equality of Means			
DENSITY	**t**	**df**	**sig (2-tailed)**
Open Forest	-0.022	73	0.983
Forest (A)	-2.765	111	0.007
Forest (B)	-0.293	200	0.77
Forest (C)	-1.911	174	0.058
Forest (D)	-1.058	100	0.293